This document is in draft form, for the purposes of soliciting feedback from independent peer review.

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List of Acronyms

Placeholder

1 APPENDIX XX—RANGE-WIDE STATUS OF THE SPECIES AND CRITICAL HABITAT

This opinion examines the status of each species that would be adversely affected by the proposed action. The status is determined by the level of extinction risk that the listed species face, based on parameters considered in documents such as recovery plans (the Central Valley Recovery Plan (National Marine Fisheries Service 2014a)), status reviews (National Marine Fisheries Service 2015, 2016a, 2016b, 2016c), and listing decisions. This informs the description of the species' likelihood of both survival and recovery. The species status section also helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02. The opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds and coastal and marine environments that make up the designated area, and discusses the current function of the essential physical and biological features that help to form that conservation value.

1.1 Sacramento River Winter-run Chinook Salmon Evolutionarily Significant Unit (ESU)

- First listed as threatened (August 4, 1989, 54 FR 32085), reclassified as endangered (January 4, 1994, 59 FR 440)
- Reaffirmed as endangered (June 28, 2005, 70 FR 37160)
- Designated critical habitat (June 16, 1993, 58 FR 33212)

The Federally listed ESU of Sacramento River winter-run Chinook salmon and designated critical habitat occurs in the action area and may be affected by the proposed action.

1.1.1 Species Listing and Critical Habitat Designation History

The Sacramento River winter-run Chinook salmon (winter-run, *Oncorhynchus tshawytscha*) ESU, currently listed as endangered, was listed as a threatened species under emergency provisions of the ESA on August 4, 1989 (54 FR 32085), and was listed as a threatened species in a final rule on November 5, 1990 (55 FR 46515). On January 4, 1994, NMFS re-classified winter-run as an endangered species (59 FR 440). NMFS concluded that winter-run in the Sacramento River warranted listing as an endangered species due to several factors, including:

(1) the continued decline and increased variability of run sizes since its first listing as a threatened species in 1989;

(2) the expectation of weak returns in future years as the result of two small year classes (1991 and 1993); and

(3) continued threats to winter-run (January 4, 1994, 59 FR 440).

On June 28, 2005, NMFS concluded that the winter-run ESU was "in danger of extinction" due to risks to the ESU's diversity and spatial structure and, therefore, continues to warrant listing as an endangered species under the ESA (70 FR 37160). In August 2011, NMFS completed a 5-year status review of five Pacific salmon ESUs, including the winter-run ESU, and determined that the species' status should again remain as endangered (August 15, 2011, 76 FR 50447). The

2011 review concluded that although the listing remained unchanged since the 2005 review, the status of the population had declined over the past five years (2005–2010) (National Marine Fisheries Service 2011c). NMFS completed another status review in May 2016 of 28 listed species of Pacific salmon, steelhead and Eulachon, which included the winter-run ESU (May 26, 2016, 81 FR 33468). The 2016 review concluded that the winter-run ESU status should remain as endangered due to drought and poor ocean conditions since 2011 that have increased the extinction risk of the winter-run ESU (National Marine Fisheries Service 2016c).

The winter-run ESU currently consists of only one population, which is confined to the upper Sacramento River (spawning below Shasta and Keswick dams) in California's Central Valley. In addition, an artificial propagation program at the Livingston Stone National Fish Hatchery (LSNFH) produces winter-run that are considered to be part of this ESU (June 28, 2005, 70 FR 37160). Most components of the winter-run life history (*e.g.*, spawning, incubation, freshwater rearing) have been compromised by the habitat blockage in the upper Sacramento River. All historical spawning and rearing habitats have been blocked since the construction of Shasta Dam in 1943. Remaining spawning and rearing areas are completely dependent on cold water releases from Shasta Dam in order to sustain the remnant population (August 4, 1989, 54 FR 32085).

NMFS designated critical habitat for winter-run Chinook salmon on June 16, 1993 (58 FR 33212).

1.1.2 Critical Habitat for Sacramento River Winter-run Chinook Salmon

Critical habitat for winter-run was designated as the following waterways, bottom and water of the waterways and adjacent riparian zones: the Sacramento River from Keswick Dam (river mile (RM) 302) to Chipps Island (RM 0) at the westward margin of the Sacramento-San Joaquin Delta (Delta); all waters from Chipps Island westward to the Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and the Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge; and all waters of San Francisco Bay north of the San Francisco-Oakland Bay Bridge from San Pablo Bay to the Golden Gate Bridge (June 16, 1993, 58 FR 33212) (see Figure 1-1). NMFS clarified that "adjacent riparian zones" are limited to only those areas above a stream bank that provide cover and shade to the near shore aquatic areas (June 16, 1993, 58 FR 33212, 33214). Although the bypasses (*e.g.,* Yolo, Sutter, and Colusa) are not currently designated critical habitat for winter-run, NMFS recognizes that they may be utilized when inundated with Sacramento River flood flows and are important rearing habitats for juvenile winter-run. Also, juvenile winter-run may use tributaries of the Sacramento River for non-natal rearing (Maslin *et al.* 1997, Pacific States Marine Fisheries Commission (PSMFC) 2014).

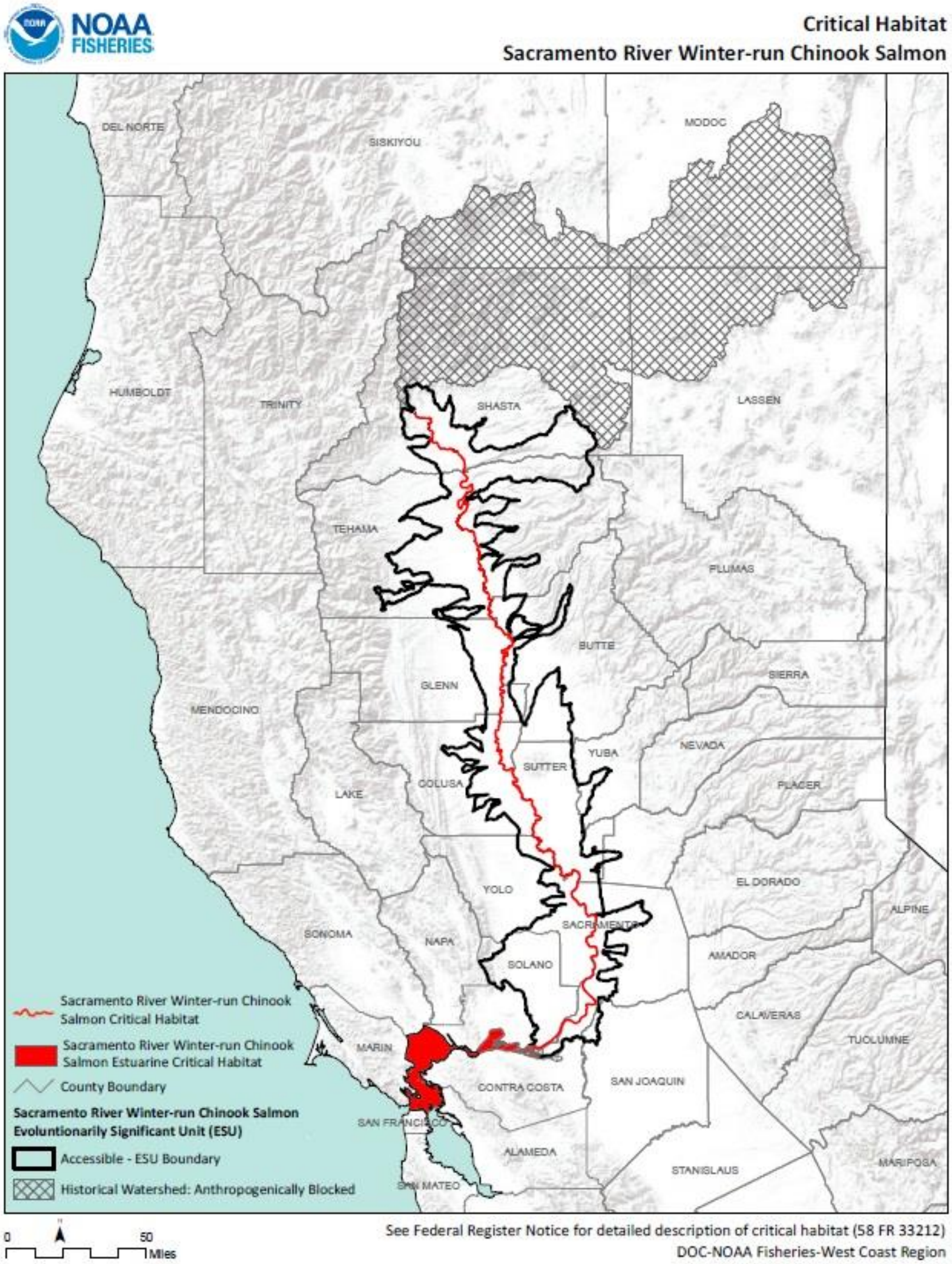


Figure 1-1. Winter-run Chinook Salmon Critical Habitat in the Central Valley.

The following subsections describe the status of the PBFs of winter-run critical habitat, which are listed in the critical habitat designation (June 16, 1993, 58 FR 33212, 33216-33217).

1.1.2.1 Adult Migration Corridors

Winter-run critical habitat PBFs include "access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River." Adult winter-run generally migrate to spawning areas during the winter and spring. At that time of year, the migration route is accessible to the appropriate spawning grounds on the upper 60 miles of the Sacramento River. Much of this migratory habitat is degraded, however, and they must pass through a fish ladder at the Anderson-Cottonwood Irrigation Dam (ACID). In addition, the many flood bypasses are known to strand adults in agricultural drains due to inadequate screening (Vincik and Johnson 2013a). Since the primary migration corridors are essential for connecting early rearing habitat with the ocean, even the degraded reaches are considered to have a high intrinsic value for the conservation of the species.

1.1.2.2 Spawning Habitat

Winter-run critical habitat PBFs include "the availability of clean gravel for spawning substrate." Suitable spawning habitat for winter-run exists in the upper 60 miles of the Sacramento River between Keswick Dam and Red Bluff Diversion Dam (RBDD) and is completely outside the historical range utilized by winter-run upstream of Keswick Dam (NMFS 2014). However, the majority of spawning habitat currently being used occurs in the first 10 miles below Keswick Dam (Stompe et al. 2016). Because Shasta and Keswick dams block gravel recruitment, the U.S. Bureau of Reclamation (Reclamation) annually injects spawning gravel into various areas of the upper Sacramento River which increases the availability of spawning substrate for a small naturally-spawning winter-run Chinook salmon population (NMFS 2016c). Even in degraded reaches, spawning habitat has a high value for the conservation of the species as its function directly affects the spawning success and reproductive potential of listed salmonids.

1.1.2.3 Adequate River Flows

Winter-run critical habitat PBFs include "adequate river flows for successful spawning, incubation of eggs, fry development and emergence, and downstream transport of juveniles." An April 5, 1960, Memorandum of Agreement between Reclamation and the California Department of Fish and Wildlife (CDFW, formerly California Department of Fish and Game) originally established flow objectives in the Sacramento River for the protection and preservation of fish and wildlife resources. In addition, Reclamation complies with the 1990 flow releases required in State Water Resource Control Board (SWRCB) Water Rights Order (WRO) 90-05 for the protection of Chinook salmon. This order includes a minimum flow release of 3,250 cubic feet per second (cfs) from Keswick Dam downstream to RBDD from September through February during all water year types, except critically dry (SWRCB 1990).

1.1.2.4 Water Temperatures

Winter-run critical habitat PBFs include "water temperatures between 42.5 and 57.5 degrees F (5.8 and 14.1 degrees C) for successful spawning, egg incubation, and fry development." Summer flow releases from Shasta Reservoir for agriculture and other consumptive uses drive operations of Shasta and Keswick dam water releases during the period of winter-run migration, spawning, egg incubation, fry development, and emergence. This flow pattern, the opposite of the pre-dam hydrograph, can provide water temperatures suitable for winter-run spawning and egg incubation for miles downstream during the hottest part of the year (Reclamation 2016). The extent to which winter-run habitat needs are met depends on Reclamation's other operational commitments, including those to water contractors, Delta requirements pursuant to State Water Rights Decision 1641 (D-1641), and Shasta Reservoir end of September storage levels required in the NMFS 2009 biological opinion on the long-term operations of the Central Valley Project and State Water Project (National Marine Fisheries Service 2009b). WRO 90-05 and 91-01 require Reclamation to operate Shasta, Keswick, and Spring Creek Powerhouse to meet a daily average water temperature of 13.3°C (56°F) at RBDD. They also provide the exception that the water temperature compliance point (TCP) may be modified when the objective cannot be met at RBDD (SWRCB 1990, SWRCB 1991). Based on these requirements, Reclamation models monthly forecasts and determines how far downstream 13.3°C (56°F) can be maintained throughout the winter-run spawning, egg incubation, and fry development stages.

In every year since WRO 90-05 and 91-1 were issued, operation plans have included modifying the TCP to make the best use of the cold water available based on water temperature modeling and current spawning distribution. Once a TCP has been identified and established in May, it generally does not change, and, therefore, water temperatures are typically adequate through the summer for successful winter-run egg incubation and fry development for those redds constructed upstream of the TCP (except for in some critically dry and drought years) (Reclamation 2016). By continually moving the TCP upstream, however, the value of that habitat is degraded by reducing the spawning area in size and imprinting upon the next generation to return further upstream.

1.1.2.5 Habitat and Adequate Prey Free of Contaminants

Winter-run critical habitat PBFs include "habitat areas and adequate prey that are not contaminated." Overall, water quality conditions in the upper Sacramento River have improved since the 1980s due to stricter standards and Environmental Protection Agency (EPA) Superfund site cleanups such as the Iron Mountain Mine. No longer are there fish kills in the Sacramento River caused by the heavy metals (*e.g.*, lead, zinc, and copper) found in the Spring Creek runoff. Legacy contaminants such as mercury (and methyl mercury), polychlorinated biphenyls, heavy metals and persistent organochlorine pesticides, however, continue to be found in watersheds throughout the Central Valley (EPA 2013). In 2010, the EPA listed the Sacramento River as impaired under Clean Water Act section 303(d), due to high levels of pesticides, herbicides, and heavy metals.

(http://www.waterboards.ca.gov/water_issues/programs/tmdl/2010state_ir_reports/category5_report.shtml)

Although most of these contaminants are at low concentrations in the food chain, they continue to work their way into the base of the food web, particularly when sediments are disturbed and previously entombed compounds are released into the water column (Cain et al. 2000).

Adequate prey for juvenile salmon to survive and grow consists of abundant aquatic and terrestrial invertebrates that make up the majority of their diet before entering the ocean. Exposure to these contaminated food sources such as invertebrates may create delayed sublethal effects that reduce fitness and survival (Laetz *et al.* 2009). Contaminants are typically associated with areas of urban development, agriculture, or other anthropogenic activities (*e.g.*, mercury contamination as a result of gold mining or processing). Freshwater rearing habitat has a high intrinsic value for the conservation of the species even if the current conditions are significantly degraded from their natural state.

1.1.2.6 Riparian and Floodplain Habitat

Winter-run critical habitat PBFs include "riparian habitat that provides for successful juvenile development and survival." The channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento River system typically have low habitat complexity, low abundance of food organisms, and offer little protection from predators. Juvenile life stages of salmonids are dependent on the natural functioning of this habitat for successful survival and recruitment. Ideal habitat contains natural cover, such as riparian canopy structure, submerged and overhanging large woody material (LWM), aquatic vegetation, large rocks and boulders, side channels, and undercut banks which augment juvenile and adult mobility, survival, and food supply. Riparian recruitment is prevented from becoming established due to the reversed hydrology (*i.e*., high summer time flows and low winter flows prevent tree seedlings from establishing). However, there are some complex, productive habitats within historical floodplains [*e.g.*, Sacramento River reaches with setback levees (*i.e.*, primarily located upstream of the City of Colusa)] and flood bypasses (*i.e.*, fish in Yolo and Sutter bypasses experience rapid growth and higher survival due to abundant food resources) seasonally available that remain in the system. Nevertheless, the current condition of degraded riparian habitat along the mainstem Sacramento River restricts juvenile growth and survival (Michel 2010, Michel *et al.* 2012).

1.1.2.7 Juvenile Emigration Corridors

Winter-run critical habitat PBFs include "access downstream so that juveniles can migrate from the spawning grounds to San Francisco Bay and the Pacific Ocean." Freshwater emigration corridors should be free of migratory obstructions, with water quantity and quality conditions that enhance migratory movements. Migratory corridors are downstream of the Keswick Dam spawning areas and include the mainstem of the Sacramento River to the Delta, as well as non-natal rearing areas near the confluence of some tributary streams.

Migratory habitat condition is strongly affected by the presence of barriers, which can include dams (*i.e.*, hydropower, flood control, and irrigation flashboard dams), unscreened or poorly screened diversions, degraded water quality, or behavioral impediments to migration. For successful survival and recruitment of salmonids, freshwater migration corridors must function sufficiently to provide adequate passage (National Marine Fisheries Service 2014). Unscreened diversions that entrain juvenile salmonids are prevalent throughout the mainstem Sacramento River and in the Delta (Herren and Kawasaki 2001). Predators such as striped bass (*Morone saxatilis*) and Sacramento pikeminnow (*Ptychocheilus grandis*) tend to concentrate immediately downstream of diversions, resulting in increased mortality of juvenile Chinook salmon (Vogel 2011).

Water pumping at the CVP/SWP export facilities in the South Delta at times causes the flow in the river to move back upstream (reverse flow), further disrupting the emigration of juvenile winter-run by attracting and diverting them to the interior Delta, where they are exposed to increased rates of predation, other stressors in the Delta, and entrainment at pumping stations. NMFS' biological opinion on the long-term operations of the CVP/SWP (National Marine

Fisheries Service 2009a) sets limits to the strength of reverse flows in the Old and Middle Rivers, thereby keeping salmon away from areas of highest mortality. Regardless of the condition, the remaining juvenile emigration corridors are of high value for the conservation of the species because they provide factors that function as rearing habitat and as an area of transition to the ocean environment.

1.1.2.8 Summary of the Physical and Biological Features of Winter-run Chinook Salmon Critical Habitat

Critical habitat for winter-run is composed of physical and biological features that are essential for the conservation of winter-run, including upstream and downstream access, and the availability of certain habitat conditions necessary to meet the biological requirements of the species. Currently, many of these physical and biological features are degraded and provide limited high quality habitat. Additional features that lessen the quality of the migratory corridor for juveniles include unscreened diversions, altered flows in the Delta, and the lack of floodplain habitat.

In addition, water operations that limit the extent of cold water below Shasta Dam have reduced the available spawning habitat (based on water temperature). Although the critical habitat for winter-run has been highly degraded, the importance of the reduced spawning habitat, migratory corridors, and rearing habitat that remains is of high value for the conservation of the species.

1.1.3 Life History

1.1.3.1 Adult Migration and Spawning

Winter-run tend to enter freshwater Winter-run exhibit a unique life history pattern (Healey 1994) compared to other salmon populations in the Central Valley (*i.e.*, spring-run, fall-run, and late-fall run) because they spawn in the summer, and the juveniles are the first to enter the ocean the following winter and spring. Adults first enter San Francisco Bay from November through June (Hallock and Fisher 1985) and migrate up the Sacramento River, past the RBDD from mid-December through early August (National Marine Fisheries Service 2014). The majority of the run passes RBDD from January through May, with the peak passage occurring in mid-March (Hallock and Fisher 1985). The timing of migration may vary somewhat due to changes in river flows, dam operations, and water year type (see Table 1-1 below; Yoshiyama *et al.* 1998, Moyle 2002b).

while still immature and travel far upriver and delay spawning for weeks or months upon arrival at their spawning grounds (Healey 1991). Spawning occurs primarily from mid-May to mid-August, with the peak activity occurring in June and July in the upper Sacramento River reach (50 miles) between Keswick Dam and RBDD (Vogel and Marine 1991). Winter-run deposit and fertilize eggs in gravel beds known as redds, which are excavated by the female who then dies following spawning. Average fecundity was 5,192 eggs/female for the 2006–2013 returns to LSNFH, which is similar to other Chinook salmon runs [*e.g.*, 5,401 average for Pacific Northwest (Quinn 2005)]. Chinook salmon spawning requirements for depth and velocities are broad, and the upper preferred water temperature is between 55–57°F (13–14°C) degrees (Snider *et al.* 2001). The majority of winter-run adults return after three years.

Table 1-1 shows the temporal occurrence of adult (a) and juvenile (b) winter-run in the Sacramento River. Darker shades indicate months of greatest relative abundance.

Table 1-1. The Temporal Occurrence of Adult (a) and Juvenile (b) Winter-run in the Sacramento River.

Winter run relative abundance	High				Medium				Low			
a) Adults freshwater												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sacramento River basin[a,b]												
Upper Sacramento River spawning[c]												
b) Juvenile emigration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sacramento River at Red Bluff [d]												
Sacramento River at Knights Landing[e]												
Sacramento trawl at Sherwood Harbor[f]												
Midwater trawl at Chipps Island[g]												

Sources: [a] (Yoshiyama *et al.* 1998); (Moyle 2002b); [b](Myers *et al.* 1998b) ; [c] (Williams 2006) ; [d] (Martin *et al.* 2001); [e] Knights Landing Rotary Screw Trap Data, CDFW (1999-2011); [f,g] Delta Juvenile Fish Monitoring Program, USFWS (1995-2012)

1.1.3.2 Egg and Fry Emergence

Winter-run incubating eggs are vulnerable to adverse effects from floods, flow fluctuations, siltation, desiccation, disease, predation during spawning, poor gravel percolation, and poor water quality. The optimal water temperature for egg incubation ranges from 46-56°F (7.8-13.3°C), and a significant reduction in egg viability occurs in mean daily water temperatures above 57.5°F (14.2°C) (Seymour 1956, Boles 1988, U.S. Fish and Wildlife Service 1999, U.S. Environmental Protection Agency 2003, Richter and Kolmes 2005, Geist *et al.* 2006).

Total embryo mortality can occur at temperatures above 62°F (16.7°C) (National Marine Fisheries Service 1997a). Depending on ambient water temperature, embryos hatch within 40-60 days and alevin (yolk-sac fry) remain in the gravel beds for an additional 4–6 weeks. As their yolk-sacs become depleted, fry begin to emerge from the gravel and start exogenous feeding in their natal stream, typically in late July to early August and continuing through October (Fisher 1994).

1.1.3.3 Juvenile Rearing and Outmigration

Juvenile winter-run have been found to exhibit variability in their life history dependent on emergence timing and growth rates (Beckman *et al.* 2007). Following spawning, egg incubation, and fry emergence from the gravel, juveniles begin to emigrate in the fall. Some juvenile winter-run migrate to sea after only 4 to 7 months of river life, while others hold and rear upstream and spend 9 to 10 months in freshwater. Emigration of juvenile winter-run fry and pre-smolts past RBDD (RM 242) may begin as early as mid-July, but typically peaks at the end of September (Table 1-1), and can continue through March in dry years (Vogel and Marine 1991, National Marine Fisheries Service 1997a).

1.1.3.4 Estuarine/Delta Rearing

Juvenile winter-run emigration into the Delta and estuary occurs primarily from November through early May based on data collected from trawls in the Sacramento River at Sherwood Harbor (West Sacramento), RM 57 (U.S. Fish and Wildlife Service 2001). The timing of emigration may vary somewhat due to changes in river flows, Shasta Dam operations, and water year type, but has been correlated with the first storm event when flows exceed 14,000 cfs at Knights Landing, RM 90, which triggers abrupt emigration towards the Delta (del Rosario *et al.* 2013). The average residence time in the Delta for juvenile winter-run is approximately 3 months based on median seasonal catch between Knights Landing and Chipps Island. In general, the earlier juvenile winter-run enter the Delta, the longer they stay and rear. Peak departure at Chipps Island regularly occurs in March (del Rosario *et al.* 2013). The Delta serves as an important rearing and transition zone for juvenile winter-run as they feed and physiologically adapt to marine waters during the smoltification process (change from freshwater to saltwater). The majority of juvenile winter-run in the Delta are 104 to 128 millimeters (mm) long based on U.S. Fish and Wildlife Service (USFWS) Delta Juvenile Fish Monitoring Program trawl data (1995-2012) and from 5 to 10 months old, by the time they depart the Delta (Fisher 1994, Myers *et al.* 1998b).

1.1.3.5 Ocean Rearing

Winter-run smolts enter the Pacific Ocean mainly in spring (March–April) and grow rapidly on a diet of small fishes, crustaceans, and squid. Salmon runs that migrate to sea at a larger size tend to have higher marine survival rates (Quinn 2005). The diet composition of Chinook salmon from California consists of anchovy, rockfish, herring, and other invertebrates, in order of preference (Healey 1991). Most Chinook from the Central Valley move northward into Oregon and Washington, where herring make up the majority of their diet. However, upon entering the ocean, winter-run tend to stay near the California coast and distribute from Point Arena southward to Monterey Bay. Winter-run have high metabolic rates, feed heavily, and grow fast compared to other fishes in their range. They can double their length and increase their weight more than ten-fold in the first summer at sea (Quinn 2005). Mortality is typically highest in the first summer at sea, but can depend on ocean conditions. Winter-run abundance has been correlated with ocean conditions, such as periods of strong up-welling, cooler temperatures, and El Nino events (Lindley *et al.* 2009a). Winter-run spend approximately 1-2 years rearing in the ocean before returning to the Sacramento River as 2-3-year-old adults. Very few winter-run Chinook salmon reach age 4. Once they reach age 3, they are large enough to become vulnerable to commercial and sport fisheries.

1.1.4 Description of Viable Salmonid Population (VSP) Parameters

As an approach to evaluate the likelihood of viability of the Sacramento River winter-run Chinook salmon ESU and determine the extinction risk of the ESU, NMFS uses the VSP concept. In this section, we evaluate the VSP parameters of abundance, productivity, spatial structure, and diversity. These specific parameters are important to consider because they are predictors of extinction risk, and the parameters reflect general biological and ecological processes that are critical to the growth and survival of salmon (McElhany *et al.* 2000b).

1.1.4.1 Abundance

Historically, winter-run population estimates were as high as 120,000 fish in the 1960s, but declined to less than 200 fish by the 1990s (National Marine Fisheries Service 2011c). In recent years, since carcass surveys began in 2001 (Figure 1-3), the highest adult escapement occurred in 2005 and 2006 with 15,839 and 17,296, respectively. However, from 2007 to 2013, the population has shown a precipitous decline, averaging 2,486 during this period, with a low of 827 adults in 2011 (Figure 1-2). This recent declining trend is likely due to a combination of factors such as poor ocean productivity (Lindley *et al.* 2009a), drought conditions from 2007-2009, low in-river survival (National Marine Fisheries Service 2011c) and extreme drought conditions in 2012-2016 (National Marine Fisheries Service 2016c). In 2015, the population was 3,015 adults, slightly above the 2007–2012 average, but below the high (17,296) for the last 10 years (California Department of Fish and Wildlife 2016).

Although impacts from hatchery fish (*i.e*., reduced fitness, weaker genetics, smaller size, less ability to avoid predators) are often cited as having deleterious impacts on natural in-river populations (Matala *et al.* 2012), the winter-run conservation program at LSNFH is strictly controlled by the USFWS to reduce such impacts. The average annual hatchery production at LSNFH is approximately 176,348 per year (2001–2010 average) compared to the estimated natural production that passes RBDD, which is 4.7 million per year based on the 2002–2010 average (Poytress and Carrillo 2011). Therefore, hatchery production typically represents approximately 3-4 percent of the total in-river juvenile production in any given year.

2014 was the third year of a drought that increased water temperatures in the upper Sacramento River, and egg-to-fry survival to the RBDD was approximately 5 percent (National Marine Fisheries Service 2016d). Due to the anticipated lower than average survival in 2014, hatchery production from LSNFH was tripled (*i.e*., 612,056 released) to offset the impact of the drought (CVP and SWP Drought Contingency Plan 2014). In 2014, hatchery production represented 83 percent of the total in-river juvenile production. In 2015, egg-to-fry survival was the lowest on record (~4 percent) due to the inability to release cold water from Shasta Dam in the fourth year of a drought. Winter-run returns in 2016 are expected to be low as they show the impact of drought on juveniles from brood year 2013 (National Marine Fisheries Service 2016d).

Figure 1-2 shows winter-run Chinook salmon escapement numbers 1967-2015, based on ladder counts and carcass surveys. After 2001 hatchery broodstock and tributaries are included, but sport catch is excluded (California Department of Fish and Wildlife 2016).

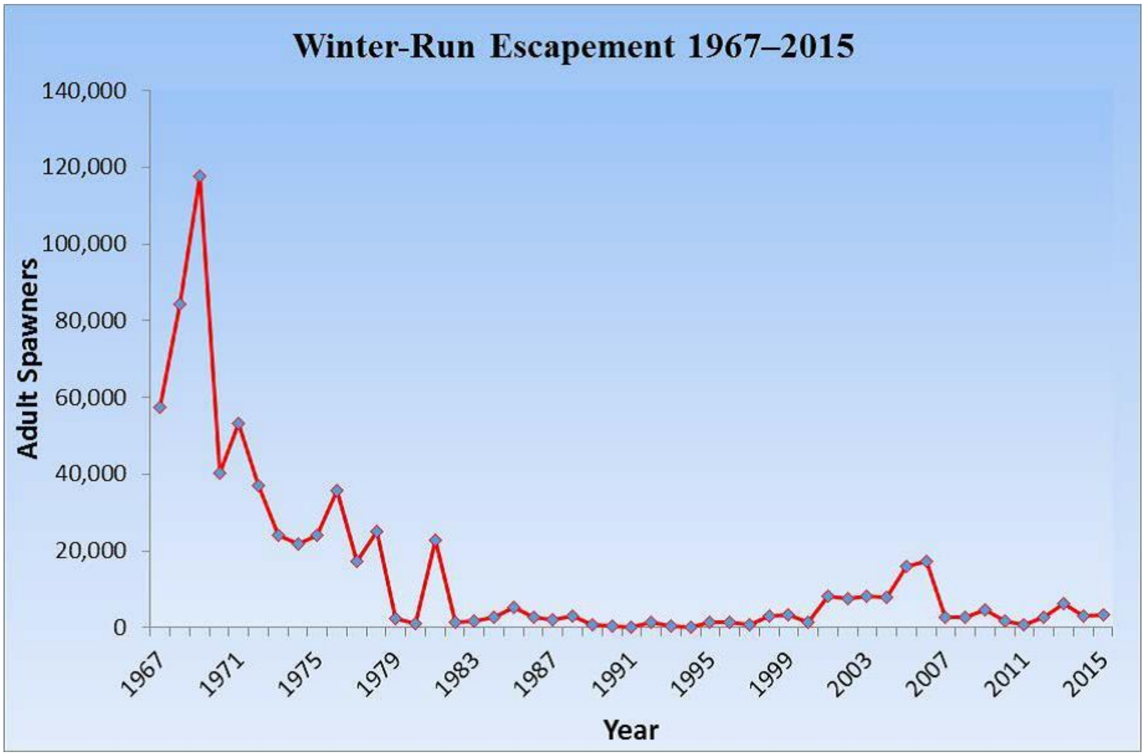


Figure 1-2. Winter-run Chinook Salmon Escapement Numbers 1967-2015

1.1.4.2 Productivity

ESU productivity was positive over 1989-2006, and adult escapement and juvenile production had been increasing annually until 2007 when productivity became negative (Figure 1-3) with declining escapement estimates. The long-term trend for the ESU, therefore, remains negative because productivity is subject to impacts from environmental and artificial conditions. The population growth rate based on cohort replacement rate (CRR) for the period 2007–2012 suggested a reduction in productivity (Figure 1-3) and indicated that the winter-run population was not replacing itself. From 2013 and 2015, winter-run experienced a positive CRR, possibly due to favorable in-river conditions in 2011 and 2012 (wet and below normal, respectively), which increased juvenile survival to the ocean.

Figure 1-3 shows winter-run population trend using cohort replacement rate derived from adult escapement, including hatchery fish, 1989–2015.

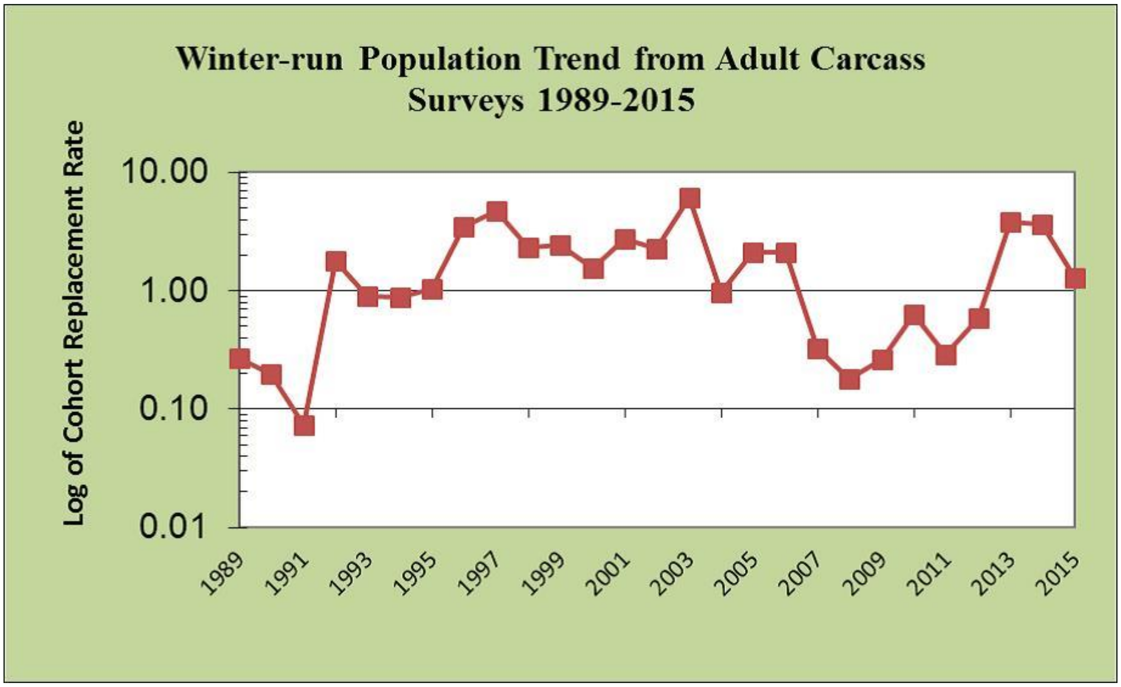


Figure 1-3. Winter-run population Trend Using Cohort Replacement Rate Derived from Adult Escapement, Including Hatchery Fish, 1989–2015

An age-structured density-independent model of spawning escapement by (Botsford and Brittnacher 1998) assessing the viability of winter-run found the species was certain to fall below the quasi-extinction threshold of three consecutive spawning runs with fewer than 50 females (Good *et al.* 2005b). Lindley and Mohr (2003) assessed the viability of the population using a Bayesian model based on spawning escapement that allowed for density dependence and a change in population growth rate in response to conservation measures. They found a biologically significant expected quasi-extinction probability of 28 percent. Although the growth rate for the winter-run population improved up until 2006, it exhibits the typical variability found in most endangered species populations. The fact that there is only one population, dependent upon cold-water releases from Shasta Dam, makes it vulnerable to periods of prolonged drought (National Marine Fisheries Service 2011c). Productivity, as measured by the number of juveniles entering the Delta, or juvenile production estimate (JPE), has declined in recent years from a high of 3.8 million in 2007 to 124,521 in 2015 (Figure 1-4). Due to uncertainties in the various JPE factors, it was updated in 2010 with the addition of confidence intervals (Cramer Fish Sciences model), and again in 2013 and 2014 with a change in survival based on acoustic tag data (National Marine Fisheries Service 2014b). However, juvenile winter-run productivity is still much lower than other Chinook salmon runs in the Central Valley and in the Pacific Northwest (Michel 2010).

Figure 1-4 shows winter-run adult and juvenile population estimates based on RBDD counts (1986–2001) and carcass counts (2001–2015). Estimates include survival to the Delta, but not through the Delta.

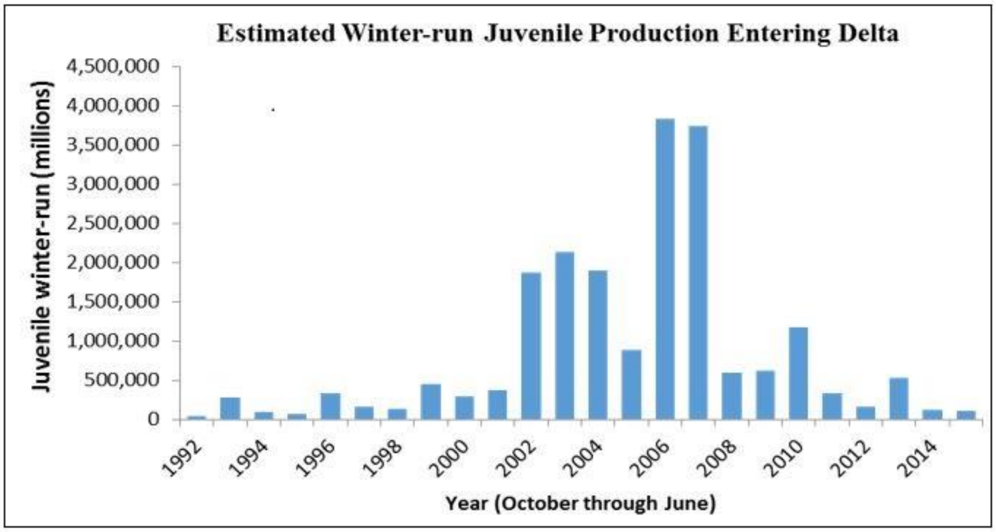


Figure 1-4. Winter-Run Adult and Juvenile Population Estimates Based on RBDD Counts (1986–2001) and Carcass Counts (2001–2015)

1.1.4.3 Spatial Structure

The distribution of winter-run spawning and initial rearing historically was limited to the upper Sacramento River (upstream of Shasta Dam), McCloud River, Pitt River, and Battle Creek, where springs provided cold water throughout the summer, allowing for spawning, egg incubation, and rearing during the mid-summer period (Yoshiyama *et al.* 1998). The construction of Shasta Dam in 1943 blocked access to all these waters except Battle Creek, which currently has its own impediments to upstream migration (*i.e.*, a number of small hydroelectric dams situated upstream of the Coleman Fish Hatchery weir). The Battle Creek Salmon and Steelhead Restoration Project (BCSSRP) is currently removing these impediments, which should restore spawning and rearing habitat for winter-run in the future. Approximately 299 miles of former tributary spawning habitat above Shasta Dam is inaccessible to winter-run. Yoshiyama *et al.* (2001) estimated that in 1938, the upper Sacramento River had a "potential spawning capacity" of approximately 14,000 redds equal to 28,000 spawners. Since 2001, the majority of winter-run redds have occurred in the first 10 miles downstream of Keswick Dam. Most components of the winter-run life history (*e.g.*, spawning, incubation, freshwater rearing) have been compromised by the construction of Shasta Dam (NMFS 2014).

The greatest risk factor for winter-run lies within its spatial structure (National Marine Fisheries Service 2011c). The remnant and remaining population cannot access 95 percent of their historical spawning habitat and must therefore be artificially maintained in the Sacramento River by:

(1) spawning gravel augmentation,

(2) hatchery supplementation, and

(3) regulation of the finite cold-water pool behind Shasta Dam to reduce water temperatures.

Winter-run require cold water temperatures in the summer that simulate their upper basin habitat, and they are more likely to be exposed to the impacts of drought in a lower basin environment. Battle Creek is currently the most feasible opportunity for the ESU to expand its spatial structure, but restoration is not scheduled to be completed until 2020. The Central Valley Salmon and Steelhead Recovery Plan includes criteria for recovering the winter-run Chinook salmon ESU, including re-establishing a population into historical habitats upstream of Shasta Dam (NMFS 2014). Additionally, National Marine Fisheries Service (2009b) included a requirement for a pilot fish passage program above Shasta Dam.

1.1.4.4 Diversity

The current winter-run population is the result of the introgression of several stocks (*e.g.*, spring-run and fall-run Chinook) that occurred when Shasta Dam blocked access to the upper watershed. A second genetic bottleneck occurred with the construction of Keswick Dam, which blocked access and did not allow spatial separation of the different runs (Good *et al.* 2005b). Lindley *et al.* (2007b) recommended reclassifying the winter-run population extinction risk from low to moderate if the proportion of hatchery origin fish from the LSNFH exceeded 15 percent due to the impacts of hatchery fish over multiple generations of spawners. Since 2005, the percentage of hatchery winter-run recovered in the Sacramento River has only been above 15 in four years: 2005, 2012, 2014, and 2015 (Figure 1-5). The average over the last 12 years (about four generations) is 13% with the most recent generation at 20% hatchery influence, putting the population at a moderate risk of extinction (NMFS 2016c).

Concern over genetic introgression within the winter-run population led to a conservation program at LSNFH that encompasses best management practices such as:

(1) genetic confirmation of each adult prior to spawning,

(2) a limited number of spawners based on the effective population size, and

(3) use of only natural-origin spawners since 2009.

These practices reduce the risk of hatchery impacts on the wild population. Hatchery-origin winter-run have made up more than 5 percent of the natural spawning run in recent years, except in 2012 when it exceeded 30 percent of the natural run (Figure 1-5). The average over the last 16 years (approximately 5 generations) has been 8 percent, which is still below the low-risk threshold (15 percent) used for hatchery influence (Lindley *et al.* (2007b). Drought conditions persisted in 2015, and hatchery production was increased again to 420,000 juveniles released, which was three times greater than what was produced naturally in-river (101,716) (CVP and SWP Drought Contingency Plan 2015).

Figure 1-5 shows percentage of hatchery-origin winter-run Chinook salmon naturally spawning in the Sacramento River (1996–2015). Source: unpublished data, (California Department of Fish and Wildlife 2016).

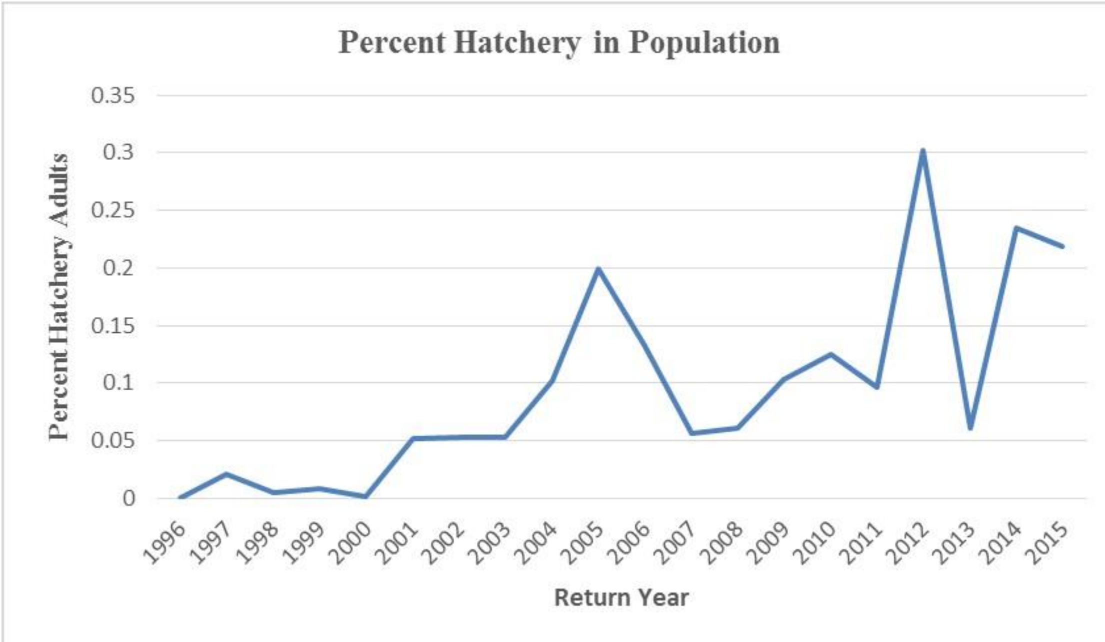


Figure 1-5. Percentage of Hatchery-Origin Winter-Run Chinook Salmon Naturally Spawning in the Sacramento River (1996–2015)

1.1.4.5 Summary of ESU Viability

There are several criteria (only one is required) that would qualify the winter-run population at moderate risk of extinction, and because there is still only one population that spawns below Keswick Dam, that winter-run ESU would be at high risk of extinction in the long-term according to criteria in Lindley *et al.* (2007b). Recent trends in those criteria are:

(1) continued low abundance (Figure 1-2);

(2) a negative growth rate over 6 years (2006–2012), which is two complete generations (Figure 1-3);

(3) a significant rate of decline since 2006;

(4) increased hatchery influence on the population (Figure 1-5); and

(5) increased risk of catastrophe from oil spills, wild fires, or extended drought (climate change).

The most recent 5-year status review (National Marine Fisheries Service 2016c) on winter-run concluded that the ESU has increased to a high risk of extinction.

In summary, the extinction risk for the winter-run ESU has increased from moderate risk to high risk of extinction since 2005, and several listing factors have contributed to the recent decline, including drought and poor ocean conditions (National Marine Fisheries Service 2016c). Large-scale fish passage and habitat restoration actions are required for improving the winter-run ESU viability (National Marine Fisheries Service 2016c).

The current condition of critical habitat for the winter-run ESU is degraded over its historical conditions. It does not provide the full extent of values for the conservation of the species necessary for the recovery of the species, particularly in the upstream riverine habitat of the

Sacramento River. Within the Sacramento River, PBFs of critical habitat (*i.e.*, migration corridor, adequate temperature, flows) have been impacted by human actions, substantially altering the historical river characteristics in which the winter-run ESU evolved. In the Delta, the man-made alterations may have a strong impact on the survival and recruitment of juvenile winter-run due to changes in migration routes and their dependence on migration cues like high flows and increased turbidity.

While some conservation measures have been successful in improving habitat conditions for the winter-run ESU since it was listed in 1989, fundamental problems with the quality of remaining habitat still remain (Cummins *et al.* 2008, Lindley *et al.* 2009b, National Marine Fisheries Service 2014c). As such, the habitat supporting this ESU remains in a highly degraded state, and it is unlikely that habitat quality has substantially changed since the last status of the species review in 2010 (National Marine Fisheries Service 2016c).

1.2 Central Valley Spring-run Chinook Salmon Evolutionarily Significant Unit (ESU)

- Listed as threatened (September 16, 1999, 64 FR 50394), reaffirmed (June 28, 2005, 70 FR 37160)
- Designated critical habitat (September 2, 2005, 70 FR 52488)

The Federally listed ESU of Central Valley (CV) spring-run Chinook salmon and designated critical habitat occurs in the action area and may be affected by the proposed action.

1.2.1 Species Listing and Critical Habitat Designation History

Central Valley (CV) spring-run Chinook salmon were originally listed as threatened on September 16, 1999 (National Marine Fisheries Service 1999) (64 FR 50394). This ESU consists of naturally spawned spring-run Chinook salmon originating from the Sacramento River basin. The Feather River Fish Hatchery (FRFH) spring-run Chinook salmon population has been included as part of the CV spring-run Chinook salmon ESU in the most recent CV spring-run Chinook salmon listing decision (National Marine Fisheries Service 2005a) (70 FR 37160, June 28, 2005). Although the FRFH spring-run Chinook salmon program is included in the ESU, the take prohibitions in 50 CFR 223.203 do not apply to these fish because they do not have an intact adipose-fin. Critical habitat was designated for CV spring-run Chinook salmon on September 2, 2005 (National Marine Fisheries Service 2005b) (70 FR 52488).

In the latest five-year review, NMFS concluded that the species' status should remain as previously listed (National Marine Fisheries Service 2016b).

1.2.2 Critical Habitat for CV Spring-run Chinook Salmon

Critical habitat for the CV spring-run Chinook salmon includes stream reaches of the Feather, Yuba, and American rivers, Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks, and the Sacramento River, as well as portions of the northern Delta. Critical habitat includes the stream channels in the designated stream reaches (September 2, 2005, 70 FR 52488).

The following subsections describe the status of the PBFs of CV spring-run Chinook salmon critical habitat, which are listed in the critical habitat designation (National Marine Fisheries Service 2005b, 70 FR 52488).

1.2.2.1 Spawning Habitat

The PBFs for CV spring-run Chinook salmon critical habitat include freshwater spawning sites with sufficient water quantity and quality conditions and substrate supporting spawning, incubation, and larval development. Most spawning habitat in the Central Valley for Chinook salmon is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation. Spawning habitat for CV spring-run Chinook salmon occurs on the mainstem Sacramento River between the Red Bluff Diversion Dam (RBDD) and Keswick Dam and in tributaries such as Mill, Deer, and Butte creeks, as well as the Feather and Yuba rivers, Big Chico, Battle, Antelope, and Clear creeks (National Marine Fisheries Service 2014). Even in degraded reaches, spawning habitat has a high value for the conservation of the species because its function directly affects the spawning success and reproductive potential of listed salmonids.

1.2.2.2 Freshwater Rearing Habitat

The PBFs for CV spring-run Chinook salmon critical habitat include freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility; water quality and forage supporting juvenile salmonid development; and natural cover such as shade, submerged and overhanging large woody material, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries also may be used for juvenile rearing. Rearing habitat condition is strongly affected by habitat complexity, food supply, and the presence of predators of juvenile salmonids (National Marine Fisheries Service 2014). Some complex, productive habitats with floodplains remain in the system (*e.g.*, the lower Cosumnes River, Sacramento River reaches with setback levees [*i.e.*, primarily located upstream of the City of Colusa]) and flood bypasses (*i.e.*, Yolo and Sutter bypasses) (Summer et al 2004, Jeffries et al 2008). However, the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system typically have low habitat complexity, low abundance of food organisms, and offer little protection from piscivorous fish and birds (National Marine Fisheries Service 2014). Freshwater rearing habitat also has a high intrinsic conservation value even if the current conditions are significantly degraded from their natural state.

1.2.2.3 Freshwater Migration Corridors

The PBFs for CV spring-run Chinook salmon critical habitat include freshwater migration corridors free of obstruction and excess predation with water quantity and quality conditions and natural cover such as submerged and overhanging large woody objects, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. Migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta. These corridors allow the upstream passage of adults and the downstream emigration of juveniles. Migratory habitat condition is strongly affected by the presence of barriers, which can include dams (*i.e.*, hydropower, flood control, and irrigation flashboard dams), unscreened or poorly screened diversions, degraded water quality, or behavioral impediments to migration (National Marine Fisheries Service 2014). For successful survival and recruitment of salmonids, freshwater migration corridors must function sufficiently to provide adequate passage. Stranding of adults has been known to occur in flood bypasses and associated weir structures (Vincik and Johnson 2013b) and a number of challenges exist on many tributary streams. For juveniles, unscreened or inadequately screened water diversions throughout their migration corridors and a scarcity of complex in-river cover have degraded this PBF (NMFS 2014). However, since the primary migration corridors are used by numerous populations, and are essential for connecting early rearing habitat with the ocean, even the degraded reaches are considered to have a high intrinsic value for the conservation of the species.

1.2.2.4 Estuarine Areas

The PBFs for CV spring-run Chinook salmon critical habitat include estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and salt water; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation (50 CFR 226.211(c)).

The remaining estuarine habitat for these species is severely degraded by altered hydrologic regimes, poor water quality, reductions in habitat complexity, and competition for food and space with exotic species. Regardless of the condition, the remaining estuarine areas are of high value for the conservation of the species because they provide factors that function to provide predator avoidance, as rearing habitat, and as an area of transition to the ocean environment.

1.2.2.5 Summary of the Physical and Biological Features of CV spring-run Chinook salmon Critical Habitat

Currently, many of the PBFs of CV spring-run Chinook salmon are degraded, and provide limited high quality habitat. Features that lessen the quality of migratory corridors for juveniles include unscreened or inadequately screened diversions, altered flows in the Delta, scarcity of complex in-river cover, and the lack of floodplain habitat. Although the current conditions of CV spring-run Chinook salmon critical habitat are significantly degraded, the spawning habitat, migratory corridors, and rearing habitat that remain are considered to have high intrinsic value for the conservation of the species.

1.2.3 Life History

1.2.3.1 Adult Migration and Holding

Chinook salmon runs are designated based on adult migration timing. Adult CV spring-run Chinook salmon leave the ocean to begin their upstream migration in late January and early February (California Department of Fish and Game 1998) and enter the Sacramento River beginning in March (Yoshiyama *et al.* 1998). Spring-run Chinook salmon move into tributaries of the Sacramento River (*e.g.*, Butte, Mill, Deer creeks) beginning as early as February in Butte Creek and typically mid-March in Mill and Deer creeks (Lindley *et al.* 2004). Adult migration peaks around mid-April in Butte Creek, and mid- to end of May in Mill and Deer creeks, and is complete by the end of July in all three tributaries [(Lindley *et al.* 2004), see Table 1-2 in text]. Typically, spring-run Chinook salmon utilize mid- to high-elevation streams that provide appropriate temperatures and sufficient flow, cover, and pool depth to allow over-summering while conserving energy and allowing their gonadal tissue to mature (Yoshiyama *et al.* 1998).

During their upstream migration, adult Chinook salmon require stream flows sufficient to provide olfactory and other orientation cues used to locate their natal streams. Adequate stream flows are necessary to allow adult passage to upstream holding habitat. The preferred temperature range for upstream migration is 3ºC (38ºF) to 13ºC (56ºF) (Bell 1990, California Department of Fish and Game 1998).

Boles (1988) recommends water temperatures below 18ºC (65ºF) for adult Chinook salmon migration, and Lindley *et al.* (2004) report that adult migration is blocked when temperatures reach 21ºC (70ºF), and that fish can become stressed as temperatures approach 21ºC (70ºF). Reclamation reports that spring-run Chinook salmon holding in upper watershed locations prefer water temperatures below 15.6ºC (60ºF); although salmon can tolerate temperatures up to 18ºC (65ºF) before they experience an increased susceptibility to disease (Williams 2006).

1.2.3.2 Adult Spawning

Spring-run Chinook salmon spawning occurs in September and October (Moyle 2002a). Chinook salmon typically mature between 2 and 6 years of age (Myers *et al.* 1998a), but primarily at age 3 (Fisher 1994). Between 56 and 87 percent of adult spring-run Chinook salmon that enter the Sacramento River basin to spawn are 3 years old (Calkins *et al.* 1940, Fisher 1994); spring-run Chinook salmon tend to enter freshwater as immature fish, migrate far upriver, and delay spawning for weeks or months.

Spring-run Chinook salmon spawning typically occurs in gravel beds that are located at the tails of holding pools (U. S. Fish and Wildlife Service 1995, National Marine Fisheries Service 2007). Spawning Chinook salmon require clean, loose gravel in swift, relatively shallow riffles or along the margins of deeper runs, and suitable water temperatures, depths, and velocities for redd construction and adequate oxygenation of incubating eggs. The range of water depths and velocities in spawning beds that Chinook salmon find acceptable is very broad. Velocity typically ranging from 1.2 feet/second to 3.5 feet/second, and water depths greater than 0.5 feet (HDR/Surface Water Resources Inc. 2007). The upper preferred water temperature for spawning Chinook salmon is 13 to 14ºC (55 to 57°F) (Chambers 1956, Smith 1973, Bjornn and Reiser 1991, California Department of Fish and Game 2001). Chinook salmon are semelparous (die after spawning).

1.2.3.3 Eggs and Fry Incubation to Emergence

The CV spring-run Chinook salmon embryo incubation period encompasses the time period from egg deposition through hatching, as well as the additional time while alevins remain in the gravel while absorbing their yolk sac before emergence. A compilation of data from multiple surveys has shown that Chinook salmon prefer a range of substrate sizes between approximately 22mm and 48mm (Kondolf and Wolman 1993). The length of time for CV spring-run Chinook salmon embryos to develop depends largely on water temperatures. In well oxygenated intergravel environs where water temperatures range from about 5 to 13ºC (41 to 55.4°F) embryos hatch in 40 to 60 days and remain in the gravel as alevins for another 4 to 6 weeks, usually after the yolk sac is fully absorbed) (National Marine Fisheries Service 2014a). In Butte and Big Chico creeks, emergence occurs from November through January, and in the colder waters of Mill and Deer creeks, emergence typically occurs from January through as late as May (Moyle 2002a). Incubating eggs require sufficient concentrations of dissolved oxygen. (Coble 1961) noted that a positive correlation exists between dissolved oxygen (DO) levels and flow within redd gravel, and Geist *et al.* (2006) observed an emergence delay of 6-10 days at 4 mg/L DO relative to water with complete oxygen saturation.

Incubating eggs are vulnerable to adverse effects from floods, siltation, desiccation, disease, predation, poor gravel permeability, and poor water quality. Studies of Chinook salmon egg survival to emergence conducted by Shelton (1955) indicated 87 percent of fry emerged successfully from large gravel with adequate subgravel flow. The optimal water temperature for egg incubation ranges from 5 to 14 ºC (41 to 56°F) (Rich 1997, Moyle 2002a). A significant reduction in egg viability occurs at water temperatures above 14ºC (57.5ºF) and total embryo mortality can occur at temperatures above 17ºC (62°F) (Myrick and Cech 2001). Alderdice and Velsen (1978) found that the upper and lower temperatures resulting in 50 percent pre-hatch mortality were 16ºC and 3ºC (61°F and 37°F), respectively, when the incubation temperature was held constant. As water temperatures increase, the rate of embryo malformations also increases, as well as the susceptibility to fungus and bacterial infestations. The length of development for Chinook salmon embryos is dependent on the ambient water temperature surrounding the redd egg pocket. Colder water necessitates longer development times as metabolic processes are slowed. Within the appropriate water temperature range for embryo incubation, embryos hatch in 40 to 60 days, and the alevins remain in the gravel for an additional 4 to 6 weeks before emerging from the gravel.

During the 4- to 6-week period when alevins remain in the gravel, they use their yolk-sac to nourish their bodies. As their yolk-sac is depleted, fry begin to emerge from the gravel to begin exogenous feeding in their natal stream. The newly emerged fry disperse to the margins of their natal stream, seeking out shallow waters with slower currents, finer sediments, and bank cover such as overhanging and submerged vegetation, root wads, and fallen woody debris, and begin feeding on zooplankton, small insects, and small invertebrates. As they switch from endogenous nourishment to exogenous feeding, the fry's yolk-sac is reabsorbed, and the belly suture closes over the former location of the yolk-sac (button-up fry). Fry typically range from 25 to 40 mm during this stage. Some fry may take up residence in their natal stream for several weeks to a year or more, while others migrate downstream to suitable habitat. Once started downstream, fry may continue downstream to the estuary and rear, or may take up residence in river reaches farther downstream for a period of time ranging from weeks to a year (Healey 1991).

1.2.3.4 Juvenile Rearing and Outmigration

Once juveniles emerge from the gravel, they initially seek areas of shallow water and low velocities while they finish absorbing the yolk sac and transition to exogenous feeding (Moyle 2002a). Many also will disperse downstream during high-flow events. As is the case in other salmonids, there is a shift in microhabitat use by juveniles to deeper faster water as they grow larger. Microhabitat use can be influenced by the presence of predators, which can force fish to select areas of heavy cover and suppress foraging in open areas (Moyle 2002a).

When juvenile Chinook salmon reach a length of 50 mm to 57 mm, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures. In the mainstems of larger rivers, juveniles tend to migrate along the margins and avoid the elevated water velocities found in the thalweg of the channel. When the channel of the river is greater than 9 to 10 feet deep, juvenile salmon tend to inhabit the surface waters (Healey 1982). Migrational cues, such as increasing turbidity from runoff, increased flows, changes in day length, or intraspecific competition from other fish in their natal streams may spur outmigration of juveniles when they have reached the appropriate stage of development (Kjelson *et al.* 1982, Brandes and McLain 2001).

As fish begin their emigration, they are displaced by the river's current downstream of their natal reaches. Similar to adult movement, juvenile salmonid downstream movement is primarily crepuscular. The daily migration of juveniles passing RBDD is highest in the four-hour period before sunrise (Martin *et al.* 2001). Juvenile Chinook salmon migration rates vary considerably depending on the physiological stage of the juvenile and hydrologic conditions. Kjelson *et al.* (1982) found that Chinook salmon fry travel as fast as 30 km per day in the Sacramento River. As Chinook salmon begin the smolt stage, they prefer to rear further downstream where ambient salinity is up to 1.5 to 2.5 parts per thousand (Healey 1979, Levy and Northcote 1981).

Spring-run Chinook salmon fry emerge from the gravel from November to March (Moyle 2002a), and the emigration timing is highly variable because they may migrate downstream as young-of-the-year, or as juveniles, or yearlings.

The modal size of fry migrants at approximately 40 mm between December and April in Mill, Butte, and Deer creeks reflects a prolonged emergence of fry from the gravel (Lindley *et al.* 2004). Studies in Butte Creek (Ward *et al.* 2003, McReynolds *et al.* 2007a) found the majority of CV spring-run Chinook salmon migrants to be fry, which emigrated primarily during December, January, and February; and that these movements appeared to be influenced by increased flow. Small numbers of CV spring-run Chinook salmon were observed to remain in Butte Creek to rear and migrated as yearlings later in the spring.

Juvenile emigration patterns in Mill and Deer creeks are very similar to patterns observed in Butte Creek, with the exception that Mill and Deer creek juveniles typically exhibit a later young-of-the-year migration and an earlier yearling migration (Lindley *et al.* 2004). The California Department of Fish and Game (1998) observed the emigration period for spring-run Chinook salmon extending from November to early May, with up to 69 percent of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period. Peak movement of juvenile CV spring-run Chinook salmon in the Sacramento River at Knights Landing occurs in December and again in March and April. However, juveniles also are observed between November and the end of May (Snider and Titus 2000).

Fry and parr may rear within riverine or estuarine habitats of the Sacramento River, the Delta, and their tributaries. Also, CV spring-run Chinook salmon juveniles have been observed rearing in the lower reaches of non-natal tributaries and intermittent streams in the Sacramento Valley during the winter months (Maslin *et al.* 1997, California Department of Fish and Game 2001). Within the Delta, juvenile Chinook salmon forage in shallow areas with protective cover, such as intertidal and subtidal mudflats, marshes, channels, and sloughs (McDonald 1960, Dunford 1975). Cladocerans, copepods, amphipods, and larvae of *Diptera*, as well as small arachnids and ants, are common prey items (Kjelson *et al.* 1982, Sommer *et al.* 2001, MacFarlane and Norton 2002). Shallow water habitats are more productive than the main river channels, supporting higher growth rates, partially due to higher prey consumption rates, as well as favorable environmental temperatures (Sommer *et al.* 2001). Optimal water temperatures for the growth of juvenile Chinook salmon in the Delta are between 12 to 14ºC (54 to 57ºF) (Brett 1952).

1.2.3.5 Estuarine Rearing

Within the estuarine habitat, juvenile Chinook salmon movements are dictated by the tidal cycles, following the rising tide into shallow water habitats from the deeper main channels and returning to the main channels when the tide recedes (Levy and Northcote 1981, Levings 1982, Levings *et al.* 1986, Healey 1991). As juvenile Chinook salmon increase in length, they tend to school in the surface waters of the main and secondary channels and sloughs, following the tides into shallow water habitats to feed (Allen and Hassler 1986). In Suisun Marsh, Moyle *et al.* (1989) reported that Chinook salmon fry tend to remain close to the banks and vegetation, near protective cover, and in dead-end tidal channels. Kjelson *et al.* (1982) reported that juvenile Chinook salmon demonstrated a diel migration pattern, orienting themselves to nearshore cover and structure during the day, but moving into more open, offshore waters at night. The fish also distributed themselves vertically in relation to ambient light. During the night, juveniles were distributed randomly in the water column, but would school up during the day into the upper 3 meters of the water column. Available data indicate that juvenile Chinook salmon use Suisun Marsh extensively both as a migratory pathway and rearing area as they move downstream to the Pacific Ocean (O'Rear and Moyle 2012).

1.2.3.6 Ocean Rearing

Once in the ocean, juvenile Chinook salmon tend to stay along the California Coast (Moyle 2002a). This is likely due to the high productivity caused by the upwelling of the California Current. These food-rich waters are important to ocean survival, as indicated by a decline in survival during years when the current does not flow as strongly and upwelling decreases (Moyle 2002a, Lindley *et al.* 2009b). After entering the ocean, juveniles become voracious predators on small fish and crustaceans, and invertebrates such as crab larvae and amphipods. As they grow larger, fish increasingly dominate their diet. They typically feed on whatever pelagic plankton is most abundant, usually herring, anchovies, juvenile rockfish, and sardines. The ocean stage of the Chinook life cycle lasts one to five years. Information on salmon abundance and distribution in the ocean is based upon CWT recoveries from ocean fisheries. For over 30 years, the marine distribution and relative abundance of specific stocks, including ESA-listed ESUs, has been estimated using a representative CWT hatchery stock (or stocks) to serve as proxies for the natural and hatchery-origin fish within ESUs. One extremely important assumption of this approach is that hatchery and natural stock components are similar in their life histories and ocean migration patterns (Knudsen et al 1999).

Ocean harvest of Central Valley Chinook salmon is estimated using an abundance index, called the Central Valley Index (CVI). The CVI is the ratio of Chinook salmon harvested south of Point Arena (where 85 percent of Central Valley Chinook salmon are caught) to escapement (adult spawner populations that have "escaped" the ocean fisheries and made it into the rivers to spawn). CWT returns indicate that Sacramento River Chinook salmon congregate off the California coast between Point Arena and Morro Bay (NMFS 2013).

Table 1-2 shows the temporal occurrence of adult (a) and juvenile (b) Central Valley spring-run Chinook salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance.

Table 1-2. The temporal occurrence of adult (a) and juvenile (b) Central Valley spring-run Chinook salmon in the Sacramento River

(a) Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a,b]												
Sac. River Mainstem[b,c]												
Mill Creek[d]												
Deer Creek[d]												
Butte Creek[d,g]												
(b) Adult Holding[a,b]												
(c) Adult Spawning[a,b,c]												
(d) Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River Tribs[e]												
Upper Butte Creek[f,g]												
Mill, Deer, Butte Creeks[d,g]												
Sac. River at RBDD[c]												
Sac. River at KL[h]												

Relative Abundance: = High = Medium = Low

Sources: [a]Yoshiyama et al. (1998); [b]Moyle (2002); [c]Myers *et al.* (1998); [d]Lindley et al. (2004); [e]CDFG (1998); [f]McReynolds et al. (2007); [g]Ward et al. (2003); [h]Snider and Titus (2000)

Note: Yearling spring-run Chinook salmon rear in their natal streams through the first summer following their birth. Downstream emigration generally occurs the following fall and winter. Most young-of-the-year spring-run Chinook salmon emigrate during the first spring after they hatch.

1.2.4 Description of Viable Salmonid Population (VSP) Parameters

As an approach to evaluate the likelihood of viability of the CV spring-run Chinook salmon ESU and determine the extinction risk of the ESU, NMFS uses the VSP concept. In this section, we evaluate the VSP parameters of abundance, productivity, spatial structure, and diversity. These specific parameters are important to consider because they are predictors of extinction risk, and the parameters reflect general biological and ecological processes that are critical to the growth and survival of salmon (McElhany *et al.* 2000b).

1.2.4.1 Abundance

Historically spring-run Chinook salmon were the second most abundant salmon run in the Central Valley and one of the largest on the west coast (California Department of Fish and Game 1990). These fish occupied the upper and middle elevation reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Stone 1872, Rutter 1904, Clark 1929).

The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s (California Department of Fish and Game 1998). The San Joaquin River historically supported a large run of spring-run Chinook salmon, suggested to be one of the largest runs of any Chinook salmon on the West Coast with estimates averaging 200,000–500,000 adults returning annually (California Department of Fish and Game 1990). Construction of Friant Dam on the San Joaquin River began in 1939 and when completed in 1942 blocked access to all upstream habitat.

The FRFH spring-run Chinook salmon population represents the only remaining evolutionary legacy of the spring-run Chinook salmon populations that once spawned above Oroville Dam, and has been included in the ESU based on its genetic linkage to the natural spawning population and the potential development of a conservation strategy for the hatchery program. On the Feather River, significant numbers of spring-run Chinook salmon, as identified by run timing, return to the FRFH. Since 1954, spawning escapement has been estimated using combinations of in-river estimates and hatchery counts, with estimates ranging from 2,908 in 1964 to two fish in 1978 (California Department of Water Resources 2001). However, after 1981, CDFG (now CDFW, California Department of Fish and Wildlife) ceased to estimate in-river spawning spring-run Chinook salmon because spatial and temporal overlap with fall-run Chinook salmon spawners made it impossible to distinguish between the two races. Spring-run Chinook salmon estimates after 1981 have been based solely on salmon entering the hatchery during the month of September. The 5-year moving averages from 1997 to 2006 had been more than 4,000 fish, but from 2007 to 2011, the 5-year moving averages have declined each year to a low of 1,742 fish in 2011, and 2012 through 2015 were back up slightly to just over 2,000 fish [(California Department of Fish and Wildlife 2016); Table 1-3].

Genetic testing has indicated that substantial introgression has occurred between fall-run and spring-run Chinook salmon populations within the Feather River system due to temporal overlap and hatchery practices (California Department of Water Resources 2001). Because Chinook salmon have not always been spatially separated in the FRFH, spring-run and fall-run Chinook salmon have been spawned together, thus compromising the genetic integrity of the spring-run Chinook salmon stock (Good *et al.* 2005a, Cavallo *et al.* 2011).

In addition, coded-wire tag (CWT) information from these hatchery returns has indicated that fall-run and spring-run Chinook salmon have overlapped, providing further evidence that the two runs have been interbred in the hatchery (California Department of Water Resources 2001). For the reasons discussed above, the FRFH spring-run Chinook salmon numbers are not included in the following discussion of ESU abundance trends.

Monitoring the Sacramento River mainstem during spring-run Chinook salmon spawning timing indicates that some spawning occurs in the river. The lack of physical separation of spring-run Chinook salmon from fall-run Chinook salmon is complicated by overlapping migration and spawning periods. Significant hybridization with fall-run Chinook salmon makes identification of spring-run Chinook salmon in the mainstem very difficult, but counts of Chinook salmon redds in September are typically used as an indicator of spring-run Chinook salmon abundance. Less than fifteen Chinook salmon redds per year were observed in the Sacramento River from 1989 to 1993, during September aerial redd counts (The Energy Planning and Instream Flow Branch 2003).

Redd surveys conducted in September between 2001 and 2011 have observed an average of 36 Chinook salmon redds from Keswick Dam downstream to the RBDD, ranging from 3 to 105 redds; 2012 observed zero redds, and 2013, 57 redds in September (California Department Fish and Wildlife, unpublished data, 2014).

Therefore, even though physical habitat conditions can support spawning and incubation, spring-run Chinook salmon depend on spatial segregation and geographic isolation from fall-run Chinook salmon to maintain genetic diversity. With the onset of fall-run Chinook salmon spawning occurring in the same time and place as potential spring-run Chinook salmon spawning, it is likely extensive introgression between the populations has occurred (California Department of Fish and Game 1998). For these reasons, Sacramento River mainstem spring-run Chinook salmon are not included in the following discussion of ESU abundance trends.

Sacramento River tributary populations in Mill, Deer, and Butte creeks are likely the best trend indicators for the CV spring-run Chinook salmon ESU as a whole because these streams contain the majority of the abundance, and are currently the only independent populations within the ESU. Generally, these streams have shown a positive escapement trend since 1991, displaying broad fluctuations in adult abundance. All tributaries combined are shown in Table 1-3, which are dominated by returns in Mill, Deer and Butte creek. Combined tributary returns from 1988 to 2015 have ranged from 1,013 in 1993 to 23,787 in 1998 (Table 1-3). Escapement numbers are dominated by Butte Creek returns (Good *et al.* 2005a), which averaged over 7,000 fish from 1995 to 2005, but then declined in years 2006 through 2011 with an average of just over 3,000 fish. During this same period, adult returns on Mill and Deer creeks have averaged over 2,000 fish total and just over 1,000 fish total, respectively. Although trends were generally positive during this time, annual abundance estimates display a high level of fluctuation, and the overall number of CV spring-run Chinook salmon remained well below estimates of historic abundance.

Additionally, in 2002 and 2003, mean water temperatures in Butte Creek exceeded 21°C for 10 or more days in July (Williams 2006). These persistent high water temperatures, coupled with high fish densities, precipitated an outbreak of *Columnaris* (*Flexibacter columnaris*) and *Ichthyophthiriasis* (*Ichthyophthirius multifiis*) diseases in the adult spring-run Chinook salmon over-summering in Butte Creek. In 2002, this contributed to a pre-spawning mortality of approximately 20 to 30 percent of the adults. In 2003, approximately 65 percent of the adults succumbed, resulting in a loss of an estimated 11,231 adult spring-run Chinook salmon in Butte Creek due to the diseases. In 2015, Butte Creek again experienced severe temperature conditions, with nearly 2,000 fish entering the creek, only 1,081 observed during the snorkel survey, and only 413 carcasses observed, which indicates a large number of pre-spawn mortality.

Declines in abundance from 2005 to 2016 placed the Mill Creek and Deer Creek populations in the high extinction risk category due to the rates of decline, and in the case of Deer Creek, also the level of escapement (National Marine Fisheries Service 2016b). Butte Creek has sufficient abundance to retain its low extinction risk classification, but the rate of population decline in years 2006 through 2016 was nearly sufficient to classify it as a high extinction risk based on this criteria. Nonetheless, the watersheds identified as having the highest likelihood of success for achieving viability/low risk of extinction include Butte, Deer, and Mill creeks (National Marine Fisheries Service 2016b). Some other tributaries to the Sacramento River, such as Clear Creek and Battle Creek, have seen population gains in the years from 2001 to 2014, but the overall abundance numbers have remained low. 2012 was a good return year for most of the tributaries with some, such as Battle Creek, having the highest return on record (799). Additionally, 2013 escapement numbers increased, in most tributary populations, which resulted in the second highest number of spring-run Chinook salmon returning to the tributaries since 1998. However, 2014 escapement numbers appear to be lower, just over 5,000 fish for the tributaries combined, which indicates a highly fluctuating and unstable ESU abundance. Even more concerning were returns for 2015, which were record lows for some populations. The next several years are anticipated to remain quite low as the effects of the 2012-2015 drought are fully realized (National Marine Fisheries Service 2016b).

1.2.4.2 Productivity

The productivity of a population (*i.e.*, production over the entire life cycle) can reflect conditions (*e.g.*, environmental conditions) that influence the dynamics of a population and determine abundance. In turn, the productivity of a population allows an understanding of the performance of a population across the landscape and habitats in which it exists and its response to those habitats (McElhany *et al.* 2000b). In general, declining productivity equates to declining population abundance. McElhany *et al.* (2000b) suggested criteria for a population’s natural productivity should be sufficient to maintain its abundance above the viable level (a stable or increasing population growth rate). In the absence of numeric abundance targets, this guideline is used. Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation.

From 1993 to 2007 the 5-year moving average of the tributary population (Mill, Deer and Butte creeks) CRR remained over 1.0, but then declined to a low of 0.47 in years 2007 through 2011 (see Table 1-3 for CV spring-run Chinook salmon population estimates with corresponding CRRs from 1986-2015). The productivity of the Feather River and Yuba River populations and contribution to the CV spring-run Chinook salmon ESU currently is unknown, however the FRFH currently produces 2,000,000 juveniles each year. The CRR for the 2012 combined tributary population was 3.84 and 8.68 in 2013, due to increases in abundance for most populations. Although 2014 returns were lower than the previous two years, the CRR was still positive (1.85). However, 2015 returns were very low, with a CRR of 0.14 when using Butte

Creek snorkel survey numbers—the lowest on record. Using the Butte Creek carcass surveys, the 2015 CRR for just Butte Creek was only 0.02.

Table 1-3. Central Valley Spring-run Chinook salmon population estimates from CDFW Grand Tab (2015) with corresponding cohort replacement rates for years since 1986.

Year	Sacramento River Basin Escapement Run Size[a]	FRFH Population	Tributary Populations	5-Year Moving Average Tributary Population Estimate	Trib CRR[b]	5-Year Moving Average of Trib CRR	5-Year Moving Average of Basin Population Estimate	Basin CRR	5-Year Moving Average of Basin CRR
1986	3,638	1,433	2,205						
1987	1,517	1,213	304						
1988	9,066	6,833	2,233						
1989	7,032	5,078	1,954		0.89			1.93	
1990	3,485	1,893	1,592	1,658	5.24		4,948	2.30	
1991	5,101	4,303	798	1,376	0.36		5,240	0.56	
1992	2,673	1,497	1,176	1,551	0.60		5,471	0.38	
1993	5,685	4,672	1,013	1,307	0.64	1.55	4,795	1.63	1.22
1994	5,325	3,641	1,684	1,253	2.11	1.79	4,454	1.04	1.18
1995	14,812	5,414	9,398	2,814	7.99	2.34	6,719	5.54	1.83
1996	8,705	6,381	2,324	3,119	2.29	2.73	7,440	1.53	2.03
1997	5,065	3,653	1,412	3,166	0.84	2.77	7,918	0.95	2.14
1998	30,533	6,746	23,787	7,721	2.53	3.15	12,888	2.06	2.23
1999	9,838	3,731	6,107	8,606	2.63	3.26	13,791	1.13	2.24
2000	9,201	3,657	5,544	7,835	3.93	2.44	12,669	1.82	1.50
2001	16,865	4,135	12,730	9,916	0.54	2.09	14,300	0.55	1.30
2002	17,212	4,189	13,023	12,238	2.13	2.35	16,730	1.75	1.46
2003	17,691	8,662	9,029	9,287	1.63	2.17	14,161	1.92	1.43
2004	13,612	4,212	9,400	9,945	0.74	1.79	14,916	0.81	1.37
2005	16,096	1,774	14,322	11,701	1.10	1.23	16,295	0.94	1.19
2006	10,828	2,061	8,767	10,908	0.97	1.31	15,088	0.61	1.21
2007	9,726	2,674	7,052	9,714	0.75	1.04	13,591	0.71	1.00
2008	6,162	1,418	4,744	8,857	0.33	0.78	11,285	0.38	0.69
2009	3,801	989	2,812	7,539	0.32	0.69	9,323	0.35	0.60
2010	3,792	1,661	2,131	5,101	0.30	0.53	6,862	0.39	0.49
2011	5,033	1,969	3,064	3,961	0.65	0.47	5,703	0.82	0.53
2012	14,724	3,738	10,986	4,747	3.91	1.10	6,702	3.87	1.16
2013	18,384	4,294	14,090	6,617	6.61	2.36	9,147	4.85	2.06
2014	8,434	2,776	5,658	7,186	1.85	2.66	10,073	1.68	2.32
2015	3,074	1,586	1,488	7,057	0.14	2.63	9,930	0.21	2.28
Median	9,775	3,616	6,159	6,541	1.97	1.89	10,220	1.00	1.46

[a] Sacramento River Basin run size is the sum of the escapement numbers from the FRFH and the tributaries.

[b] Abbreviations: CRR = Cohort Replacement Rate, Trib = tributary

1.2.4.3 Spatial Structure

Spatial structure refers to the arrangement of populations across the landscape, the distribution of spawners within a population, and the processes that produce these patterns. Species with a restricted spatial distribution and few spawning areas are at a higher risk of extinction from catastrophic environmental events (*e.g.*, a single landslide) than are species with more widespread and complex spatial structure. Species or population diversity concerns the phenotypic (morphology, behavior, and life-history traits) and genotypic (DNA) characteristics of populations. Phenotypic diversity allows more populations to use a wider array of environments and protects populations against short-term temporal and spatial environmental changes. Genotypic diversity, on the other hand, provides populations with the ability to survive long-term changes in the environment. To meet the objective of representation and redundancy, diversity groups need to contain multiple populations to survive in a dynamic ecosystem subject to unpredictable stochastic events, such as pyroclastic events or wild fires (McElhany et al 2000).

The Central Valley Technical Review Team (TRT) estimated that historically there were 18 or 19 independent populations of CV spring-run Chinook salmon, along with a number of dependent populations, all within four distinct geographic regions, or diversity groups (Figure 1-6) (Lindley *et al.* 2004). Of these populations, only three independent populations currently exist (Mill, Deer, and Butte creeks tributary to the upper Sacramento River) and they represent only the northern Sierra Nevada diversity group. Additionally, smaller populations are currently persisting in Antelope and Big Chico creeks and the Feather and Yuba rivers in the northern Sierra Nevada diversity group (California Department of Fish and Game 1998). All historical populations in the basalt and porous lava diversity group and the southern Sierra Nevada diversity group have been extirpated, except Battle Creek in the basalt and porous lava diversity group has had a small persistent population since 1995, and the upper Sacramento River may have a small persisting population spawning in the mainstem-river as well. The northwestern California diversity group did not historically contain independent populations and currently contains two small persisting populations, in Clear Creek and Beegum Creek (tributary to Cottonwood Creek), that are likely dependent on the northern Sierra Nevada diversity group populations for their continued existence. Construction of low elevation dams in the foothills of the Sierras on the San Joaquin, Mokelumne, Stanislaus, Tuolumne, and Merced rivers, has been thought to have extirpated CV spring-run Chinook salmon from these watersheds of the San Joaquin River, as well as on the American River of the Sacramento River basin. However, observations in the last decade suggest that perhaps spring-running populations may currently occur in the Stanislaus and Tuolumne rivers (Franks 2014a).

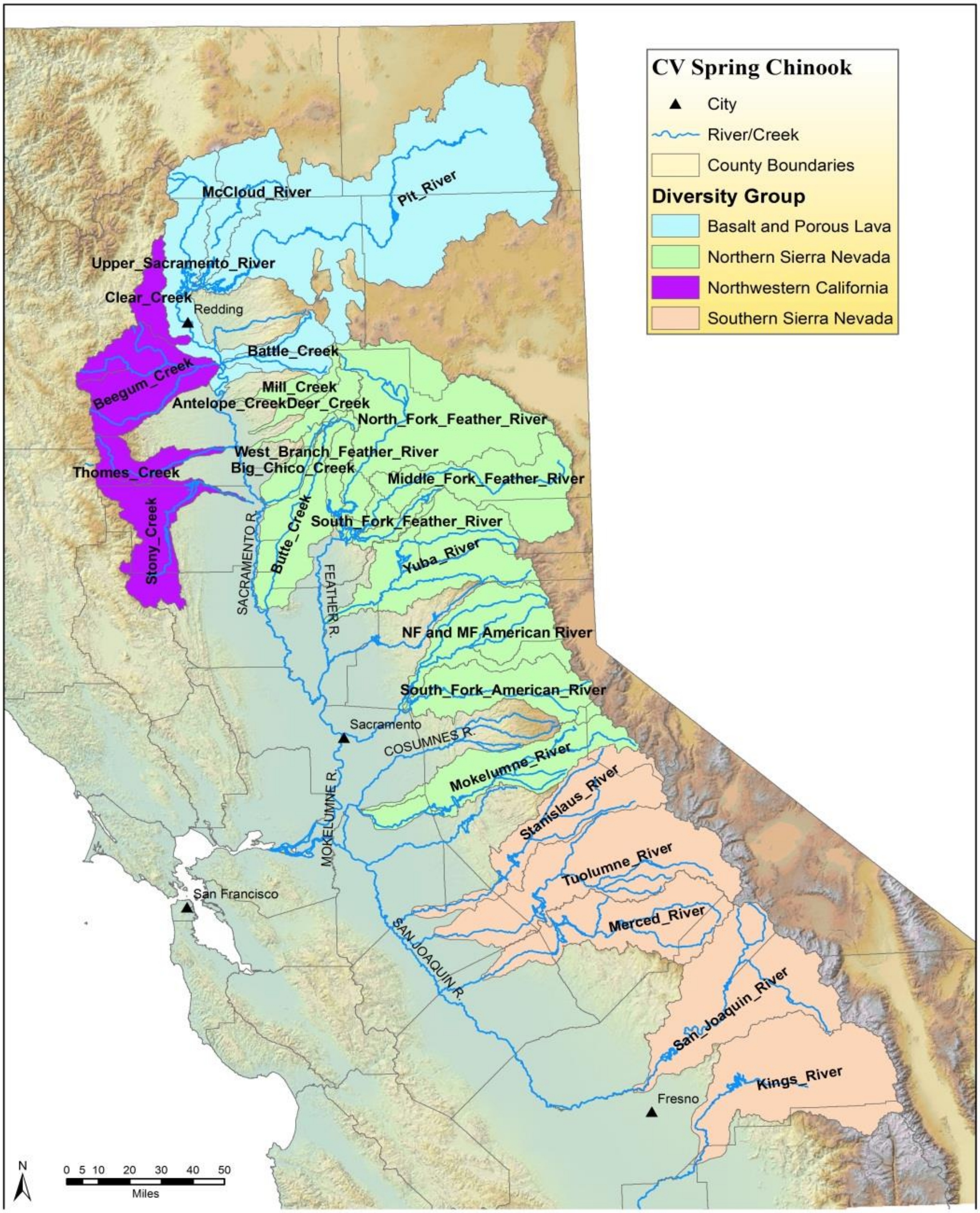


Figure 1-6 Diversity Groups for the Central Valley spring-run Chinook salmon ESU.

With only one of four diversity groups currently containing viable independent populations, the spatial structure of CV spring-run Chinook salmon is severely reduced. Butte Creek spring-run Chinook salmon adult returns are currently utilizing all available habitat in the creek; and it is unknown if individuals have opportunistically migrated to other systems. The persistent populations in Clear Creek and Battle Creek, with habitat restoration projects completed and more underway, are anticipated to add to the spatial structure of the CV spring-run Chinook salmon ESU if they can reach viable status in the basalt and porous lava and northwestern California diversity group areas. The spatial structure of the spring-run Chinook salmon ESU would still be lacking due to the extirpation of all San Joaquin River basin spring-run Chinook salmon populations; however, recent information suggests that perhaps a self-sustaining population of spring-run Chinook salmon is occurring in some of the San Joaquin River tributaries, most notably the Stanislaus and the Tuolumne rivers.

A final rule was published to designate a nonessential experimental population of CV spring-run Chinook salmon in the San Joaquin River from Friant Dam downstream to its confluence with the Merced River to allow reintroduction of the species below Friant Dam as part of the San Joaquin River Restoration Program (SJRRP) (78 FR 79622, December 31, 2013). Pursuant to ESA section 10(j), with limited exceptions, each member of an experimental population shall be treated as a threatened species. However, the rule includes protective regulations under ESA section 4(d) that provide specific exceptions to prohibitions for taking CV spring-run Chinook salmon within the experimental population area, and in specific instances elsewhere. The first release of CV spring-run Chinook salmon juveniles into the San Joaquin River occurred in April 2014. A second release occurred in 2015, and future releases are planned to continue annually during the spring. The 2016 release will include the first generation of spring-run Chinook salmon reared entirely in the San Joaquin River in over 60 years. The nonessential experimental population's contribution to the viability of the CV spring-run Chinook salmon ESU will be determined in future status assessments.

Snorkel surveys (Kennedy and Cannon 2005) conducted between October 2002 and October 2004 on the Stanislaus River identified adults in June 2003 and 2004, as well as observed Chinook fry in December 2003, which would indicate spring-run Chinook salmon spawning timing. In addition, monitoring on the Stanislaus since 2003 and on the Tuolumne since 2009, has indicated upstream migration of adult spring-run Chinook salmon (Anderson *et al.* 2007), and 114 adult were counted on the video weir on the Stanislaus River between February and June in 2013 with only 7 individuals without adipose fins (FISHBIO 2015).

Finally, rotary screw trap (RST) data provided by Stockton U.S. Fish and Wildlife Service (USFWS) corroborates the spring-run Chinook salmon adult timing by indicating that there are a small number of fry migrating out of the Stanislaus and Tuolumne at a period that would coincide with spring-run juvenile emigration (Franks 2014a). Although there have been observations of springtime running Chinook salmon returning to the San Joaquin tributaries in recent years, there is insufficient information to determine the specific origin of these fish and whether or not they are straying into the basin or returning to natal streams. Genetic assessment or natal stream analyses of hard tissues could inform our understanding of the relationship of these fish to the ESU.

Lindley *et al.* (2007a) described a general criteria for "representation and redundancy" of spatial structure, which was for each diversity group to have at least two viable populations. More specific recovery criteria for the spatial structure of each diversity group have been laid out in the NMFS Central Valley Salmon and Steelhead Recovery Plan (National Marine Fisheries Service 2014a). According to the criteria, one viable population in the Northwestern California diversity group, two viable populations in the basalt and porous lava diversity group, four viable populations in the northern Sierra Nevada diversity group, and two viable populations in the southern Sierra Nevada diversity group, in addition to maintaining dependent populations, are needed for recovery. It is clear that further efforts will need to involve more than restoration of currently accessible watersheds to make the ESU viable. The NMFS Central Valley Salmon and Steelhead Recovery Plan calls for reestablishing populations into historical habitats currently blocked by large dams, such as the reintroduction of a population upstream of Shasta Dam, and to facilitate passage of fish upstream of Englebright Dam on the Yuba River (National Marine Fisheries Service 2014a).

1.2.4.4 Diversity

Diversity, both genetic and behavioral, is critical to success in a changing environment. Salmonids express variation in a suite of traits, such as anadromy, morphology, fecundity, run timing, spawn timing, juvenile behavior, age at smolting, age at maturity, egg size, developmental rate, ocean distribution patterns, male and female spawning behavior, and physiology and molecular genetic characteristics (including rate of gene-flow among populations). Criteria for the diversity parameter are that human-caused factors should not alter variation of traits. The more diverse these traits (or the more these traits are not restricted), the more adaptable a population is, and the more likely that individuals, and therefore the species, would survive and reproduce in the face of environmental variation (McElhany *et al.* 2000b). However, when this diversity is reduced due to loss of entire life history strategies or to loss of habitat used by fish exhibiting variation in life history traits, the species is in all probability less able to survive and reproduce given environmental variation.

The CV spring-run Chinook salmon ESU is comprised of two known genetic complexes. Analysis of natural and hatchery spring-run Chinook salmon stocks in the Central Valley indicates that the northern Sierra Nevada diversity group spring-run Chinook salmon populations in Mill, Deer, and Butte creeks retain genetic integrity as opposed to the genetic integrity of the Feather River population, which has been somewhat compromised. The Feather River spring-run Chinook salmon have introgressed with the Feather River fall-run Chinook salmon, and it appears that the Yuba River spring-run Chinook salmon population may have been impacted by FRFH fish straying into the Yuba River (and likely introgression with wild Yuba River fall-run has occurred) (Garza et al 2007). Additionally, the diversity of the spring-run Chinook salmon ESU has been further reduced with the loss of the majority, if not all, of the San Joaquin River basin spring-run Chinook salmon populations. Efforts underway, such as the San Joaquin River Restoration Project to reintroduce a spring-run population below Friant Dam, are needed to improve the diversity of CV spring-run Chinook salmon (NMFS 2014).

1.2.4.5 Summary of ESU Viability

Because the populations in Butte, Deer and Mill creeks are the best trend indicators for ESU viability, we can evaluate risk of extinction based on VSP parameters in these watersheds. Lindley *et al.* (2007a) indicated that the spring-run Chinook salmon populations in the Central Valley had a low risk of extinction in Butte and Deer creeks, according to their population viability analysis (PVA) model and other population viability criteria (*i.e.*, population size, population decline, catastrophic events, and hatchery influence, which correlate with VSP parameters abundance, productivity, spatial structure, and diversity). The Mill Creek population of spring-run Chinook salmon was at moderate extinction risk according to the PVA model, but appeared to satisfy the other viability criteria for low-risk status. However, the CV spring-run Chinook salmon ESU failed to meet the "representation and redundancy rule" since there are only demonstrably viable populations in one diversity group (northern Sierra Nevada) out of the three diversity groups that historically contained them, or out of the four diversity groups as described in the NMFS Central Valley Salmon and Steelhead Recovery Plan. Over the long term, these three remaining populations are considered to be vulnerable to catastrophic events, such as volcanic eruptions from Mount Lassen or large forest fires due to the close proximity of their headwaters to each other. Drought is also considered to pose a significant threat to the viability of the spring-run Chinook salmon populations in these three watersheds due to their close proximity to each other. One large event could eliminate all three populations.

Until 2012, the status of CV spring-run Chinook salmon ESU had deteriorated on balance since the 2005 status review and the Lindley *et al.* (2007a) assessment, with two of the three extant independent populations (Deer and Mill creeks) of spring-run Chinook salmon slipping from low or moderate extinction risk to high extinction risk. Additionally, Butte Creek remained at low risk, although it was on the verge of moving towards high risk, due to rate of population decline. In contrast, spring-run Chinook salmon in Battle and Clear creeks had increased in abundance since 1998, reaching levels of abundance that place these populations at moderate extinction risk. Both of these populations have likely increased at least in part due to extensive habitat restoration. The Southwest Fisheries Science Center concluded in their viability report that the status of CV spring-run Chinook salmon ESU has probably deteriorated since the 2005 status review and that its extinction risk has increased (Williams *et al.* 2011). The degradation in status of the three formerly low- or moderate-risk independent populations is cause for concern.

The viability assessment of CV spring-run Chinook salmon conducted during NMFS' 2010 status review (National Marine Fisheries Service 2011a), found that the biological status of the ESU had worsened since the last status review (2005) and recommend that its status be reassessed in two to three years as opposed to waiting another five years, if the decreasing trend continued and the ESU did not respond positively to improvements in environmental conditions and management actions. In 2012 and 2013, most tributary populations increased in returning adults, averaging over 13,000. However, 2014 returns were lower again, just over 5,000 fish, indicating the ESU remains highly fluctuating. The most recent status review was conducted in 2015 (National Marine Fisheries Service 2016b), which looked at promising increasing populations in 2012-2014. However the 2015 returning fish were extremely low (1,488), with additional pre-spawn mortality reaching record lows. Because the effects of the 2012-2015 drought have not been fully realized, we anticipate at least several more years of very low returns, which may reach severe rates of decline (National Marine Fisheries Service 2016b).

In summary, the extinction risk for the CV spring-run Chinook salmon ESU remains at moderate risk of extinction (National Marine Fisheries Service 2016b). Based on the severity of the drought and the low escapements as well as increased pre-spawn mortality in Butte, Mill, and Deer creeks in 2015, there is concern that these CV spring-run Chinook salmon strongholds will deteriorate into high extinction risk in the coming years based on the population size or rate of decline criteria (National Marine Fisheries Service 2016b).

1.3 California Central Valley Steelhead Distinct Population Segment (DPS)

- Originally listed as threatened (March 19, 1998, 63 FR 13347), reaffirmed as threatened (January 5, 2006, 71 FR 834)
- Critical habitat designated (September 2, 2005, 70 FR 52488)

The Federally listed DPS of California Central Valley (CCV) steelhead and designated critical habitat occurs in the action area and may be affected by the proposed action.

1.3.1 Species Listing and Critical Habitat Designation History

CCV steelhead were originally listed as threatened on March 19, 1998 (63 FR 13347). Following a new status review (Good *et al.* 2005a) and after application of the agency's hatchery listing policy, NMFS reaffirmed the status of CCV steelhead as threatened and also listed the Feather River Fish Hatchery and Coleman National Fish Hatchery artificial propagation programs as part of the DPS on January 5, 2006 (71 FR 834). In doing so, NMFS applied the DPS policy to the species because the resident and anadromous life forms of *O. mykiss* remain "markedly separated" as a consequence of physical, ecological, and behavioral factors, and may therefore warrant delineation as separate DPSs (January 5, 2006, 71 FR 834). On May 5, 2016, NMFS completed another 5-year status review of CCV steelhead and recommended that the CCV steelhead DPS remain classified as a threatened species (National Marine Fisheries Service 2016c). Critical habitat was designated for CCV steelhead on September 2, 2005 (70 FR 52488).

1.3.2 Critical Habitat and Physical and Biological Features for CCV Steelhead

Critical habitat for CCV steelhead includes stream reaches such as those of the Sacramento, Feather, and Yuba Rivers, and Deer, Mill, Battle, and Antelope creeks in the Sacramento River basin; the San Joaquin River, including its tributaries, and the waterways of the Delta (Figure 1-7). Currently the CCV steelhead DPS and critical habitat extends up the San Joaquin River to the confluence with the Merced River. Critical habitat includes the stream channels in the designated stream reaches and the lateral extent as defined by the ordinary high-water line. In areas where the ordinary high-water line has not been defined, the lateral extent will be defined by the bankfull elevation (defined as the level at which water begins to leave the channel and move into the floodplain; it is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series) (Bain and Stevenson 1999) (September 2, 2005, 70 FR 52488). The following subsections describe the status of the PBFs of CCV steelhead critical habitat, which are listed in the critical habitat designation (September 2, 2005, 70 FR 52488).

1.3.2.1 Spawning Habitat

The PBFs of CCV steelhead critical habitat include freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, egg incubation, and larval development. Most of the available spawning habitat for steelhead in the Central Valley is located in areas directly downstream of dams due to inaccessibility to historical spawning areas upstream and the fact that dams are typically built at high gradient locations. These reaches are often impacted by the upstream impoundments, particularly over the summer months, when high temperatures can have adverse effects upon salmonids spawning and rearing below the dams (National Marine Fisheries Service 2014). Even in degraded reaches, spawning habitat has a high value for the conservation of the species as its function directly affects the spawning success and reproductive potential of listed salmonids.

1.3.2.2 Freshwater Rearing Habitat

The PBFs of CCV steelhead critical habitat include freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large woody material (LWM), log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries also may be used for juvenile rearing. Rearing habitat condition is strongly affected by habitat complexity, food supply, and the presence of predators of juvenile salmonids (National Marine Fisheries Service 2014). Some complex, productive habitats with floodplains remain in the system (*e.g.*, the lower Cosumnes River, Sacramento River reaches with setback levees [*i.e.*, primarily located upstream of the City of Colusa]) and flood bypasses (*i.e.*, Yolo and Sutter bypasses) (Summer et al 2004, Jeffries 2008). However, the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators (National Marine Fisheries Service 2014). Freshwater rearing habitat also has a high value for the conservation of the species even if the current conditions are significantly degraded from their natural state. Juvenile life stages of salmonids are dependent on the function of this habitat for successful survival and recruitment.

1.3.2.3 Freshwater Migration Corridors

The PBFs of CCV steelhead critical habitat include freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large woody objects, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. Migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta. These corridors allow the upstream and downstream passage of adults, and the emigration of smolts. Migratory habitat condition is strongly affected by the presence of barriers, which can include dams (*i.e.*, hydropower, flood control, and irrigation flashboard dams), unscreened or poorly screened diversions, degraded water quality, or behavioral impediments to migration (National Marine Fisheries Service 2014). For successful survival and recruitment of salmonids, freshwater migration corridors must function sufficiently to provide adequate passage. Stranding of adults has been known to occur in flood bypasses and associated weir structures (Vincik and Johnson 2013) and a number of challenges exist on many tributary streams. For juveniles, unscreened or complex in-river cover have degraded this PBF (National Marine Fisheries Service 2014). However, since the primary freshwater migration corridors are used by numerous listed fish populations, and are essential for connecting early rearing habitat with the ocean, even the degraded reaches are considered to have a high intrinsic value for the conservation of the species.

1.3.2.4 Estuarine Areas

The PBFs for CCV steelhead critical habitat include estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and salt water; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation (50 CFR 226.211(c)).

The remaining estuarine habitat for this species is severely degraded by altered hydrologic regimes, poor water quality, reductions in habitat complexity, and competition for food and space with exotic species (National Marine Fisheries Service 2014). Regardless of the conditions, the remaining estuarine areas are considered to have a high value for the conservation of the species because they provide features that function to provide predator avoidance, as rearing habitat, and as a transitional zone to the ocean environment.

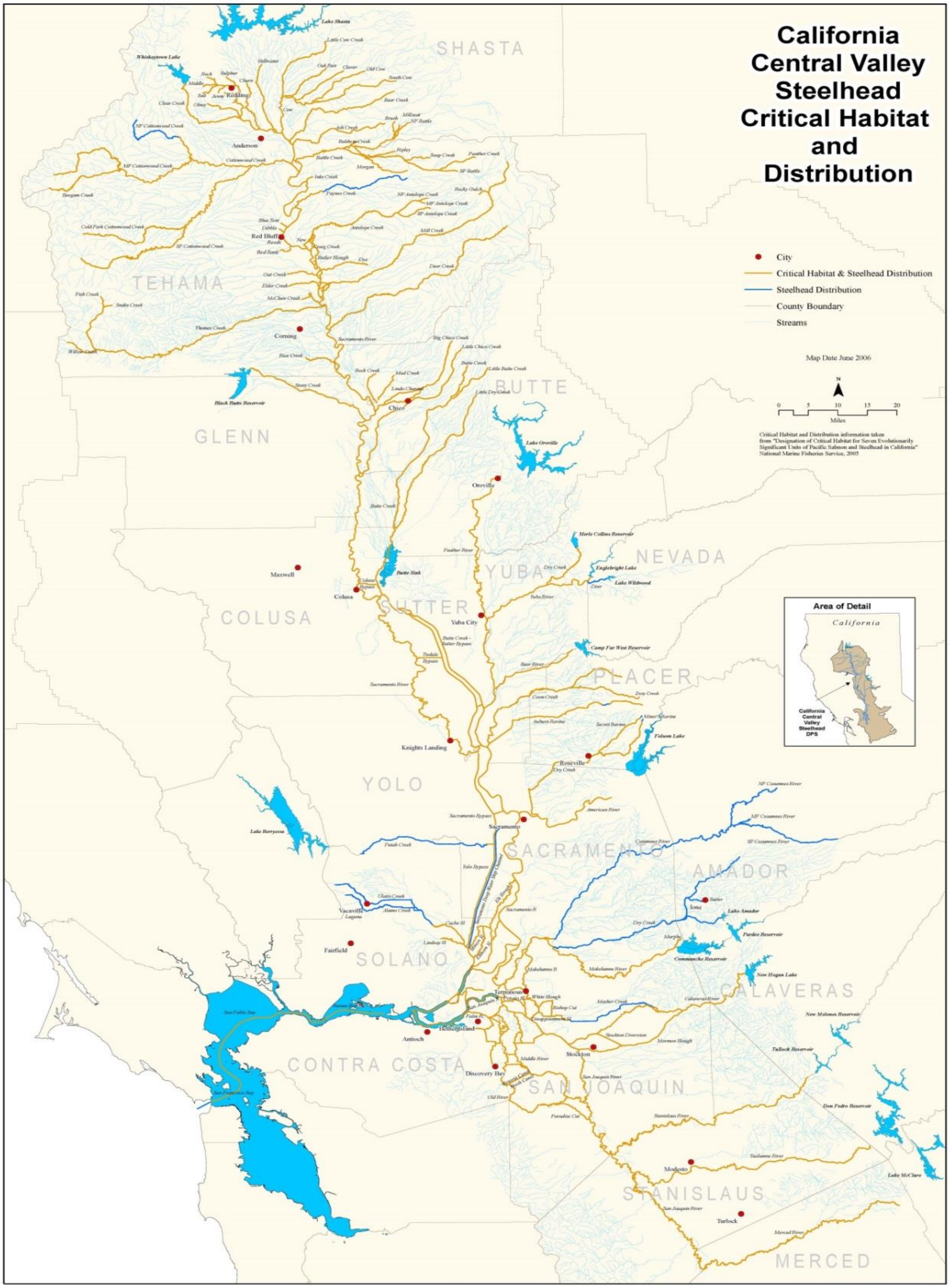


Figure 1-7. California Central Valley steelhead designated critical habitat

1.3.3 Life History

1.3.3.1 Egg to Parr

The length of time it takes for eggs to hatch depends mostly on water temperature. Steelhead eggs hatch in three to four weeks at 10°C (50°F) to 15°C (59°F) (Moyle 2002a). After hatching, alevins remain in the gravel for an additional two to five weeks while absorbing their yolk sacs, and emerge in spring or early summer (Barnhart 1986). A compilation of data from multiple surveys has shown that steelhead prefer a range of substrate sizes between approximately 18 and 35mm (Kondolf and Wolman 1993). Fry emerge from the gravel usually about four to six weeks after hatching, but factors such as redd depth, gravel size, siltation, and temperature can speed or retard this time (Shapovalov and Taft 1954). Coble (1961) noted that a positive correlation exists between dissolved oxygen levels and flow within redd gravel, and Rombough (1988) observed a critical threshold for egg survival between 7.5 and 9.7 mg/L. Upon emergence, fry inhale air at the stream surface to fill their air bladders, absorb the remains of their yolks in the course of a few days, and start to feed actively, often in schools (Barnhart 1986, National Marine Fisheries Service 1996).

The newly emerged juveniles move to shallow, protected areas associated within the stream margin (McEwan and Jackson 1996). As steelhead parr increase in size and their swimming abilities improve, they increasingly exhibit a preference for higher velocity and deeper mid-channel areas (Hartman 1965, Everest and Chapman 1972, Fontaine 1988). Growth rates have been shown to be variable and are dependent on local habitat conditions and seasonal climate patterns (Hayes *et al.* 2008).

Productive juvenile rearing habitat is characterized by complexity, primarily in the form of cover, which can be deep pools, woody debris, aquatic vegetation, or boulders. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Meehan and Bjornn 1991). Optimal water temperatures for growth range from 15°C (59°F) to 20°C (68°F) (McCullough *et al.* 2001, Spina *et al.* 2006). Cherry *et al.* (1975) found preferred temperatures for rainbow trout ranged from 11°C (51.8°F) to 21°C (69.8°F) depending on acclimation temperatures (Myrick and Joseph J. Cech 2001).

1.3.3.2 Smolt Migration

Juvenile steelhead will often migrate downstream as parr in the summer or fall of their first year of life, but this is not a true smolt migration (Loch *et al.* 1988). Smolt migrations occur in the late winter through spring, when juveniles have undergone a physiological transformation to survive in the ocean, and become slender in shape, bright silvery in coloration, with no visible parr marks. Emigrating steelhead smolts use the lower reaches of the Sacramento River and the Delta primarily as a migration corridor to the ocean. Some rearing behavior is thought to occur in tidal marshes, non-tidal freshwater marshes, and other shallow water habitats in the Delta before the fish enter the ocean (National Marine Fisheries Service 2014a).

1.3.3.3 Ocean Behavior

Unlike Pacific salmon, steelhead do not appear to form schools in the ocean (Behnke 1992). Steelhead in the southern part of their range appear to migrate close to the continental shelf, while more northern populations may migrate throughout the northern Pacific Ocean (Barnhart

1986). It is possible that California steelhead may not migrate to the Gulf of Alaska region of the North Pacific as commonly as more northern populations such as those in Washington and British Colombia. Burgner (1993) reported that no coded-wire tagged steelhead from California hatcheries were recovered from the open ocean surveys or fisheries that were sampled for steelhead between 1980 and 1988. Only a small number of disk-tagged fish from California were captured. This behavior might explain the small average size of Central Valley steelhead relative to populations in the Pacific Northwest, as food abundance in the nearshore coastal zone may not be as high as in the Gulf of Alaska.

Pearcy *et al.* (1990) found that the diets of juvenile steelhead caught in coastal waters of Oregon and Washington were highly diverse and included many species of insects, copepods, and amphipods, but by biomass the dominant prey items were small fishes (including rockfish and greenling) and euphausids.

There are no commercial fisheries for steelhead in California, Oregon, or Washington, with the exception of some tribal fisheries in Washington waters.

1.3.3.4 Spawning

CCV steelhead generally enter freshwater from August to November (with a peak in September (Hallock *et al.* 1961)), and spawn from December to April, with a peak in January through March, in rivers and streams where cold, well oxygenated water is available [Table 1-2; (Hallock *et al.* 1961, McEwan and Jackson 1996, Williams 2006). The timing of upstream migration is correlated with high flow events, such as freshets, and the associated change in water temperatures (Workman *et al.* 2002). Adults typically spend a few months in freshwater before spawning (Williams 2006), but very little is known about where they hold between entering freshwater and spawning in rivers and streams. The threshold of a 56°F maximum water temperature that is commonly used for Chinook salmon is often extended to steelhead, but temperatures for spawning steelhead are not usually a concern as this activity occurs in the late fall and winter months when water temperatures are low. Female steelhead construct redds in suitable gravel and cobble substrate, primarily in pool tailouts and heads of riffles.

Few direct counts of fecundity are available for CCV steelhead populations, but because the number of eggs laid per female is highly correlated with adult size, adult size can be used to estimate fecundity with reasonable precision. Adult steelhead size depends on the duration of and growth rate during their ocean residency (Meehan and Bjornn 1991). CCV steelhead generally return to freshwater after one or two years at sea (Hallock *et al.* 1961), and adults typically range in size from two to twelve pounds (Reynolds *et al.* 1993). Steelhead about 55 cm (FL) long may have fewer than 2,000 eggs, whereas steelhead 85 cm (FL) long can have 5,000 to 10,000 eggs, depending on the stock (Meehan and Bjornn 1991). The average for Coleman National Fish Hatchery since 1999 is about 3,900 eggs per female (U.S. Fish and Wildlife Service 2011).

Unlike Pacific salmon, steelhead are iteroparous, meaning they are capable of spawning multiple times before death (Busby *et al.* 1996). However, it is rare for steelhead to spawn more than twice before dying; and repeat spawners tend to be biased towards females (Busby *et al.* 1996). Iteroparity is more common among southern steelhead populations than northern populations (Busby *et al.* 1996). Although one-time spawners are the great majority, Shapovalov and Taft (1954) reported that repeat spawners were relatively numerous (17.2 percent) in Waddell Creek. Null (2013) found between 36 percent and 48 percent of kelts released from Coleman NFH in

2005 and 2006 survived to spawn the following spring, which is in sharp contrast to what Hallock (1989) reported for Coleman NFH in the 1971 season, where only 1.1 percent of adults were fish that had been tagged the previous year. Most populations have never been studied to determine the percentage of repeat spawners. Hatchery steelhead are typically less likely than wild fish to survive to spawn a second time (Leider *et al.* 1986).

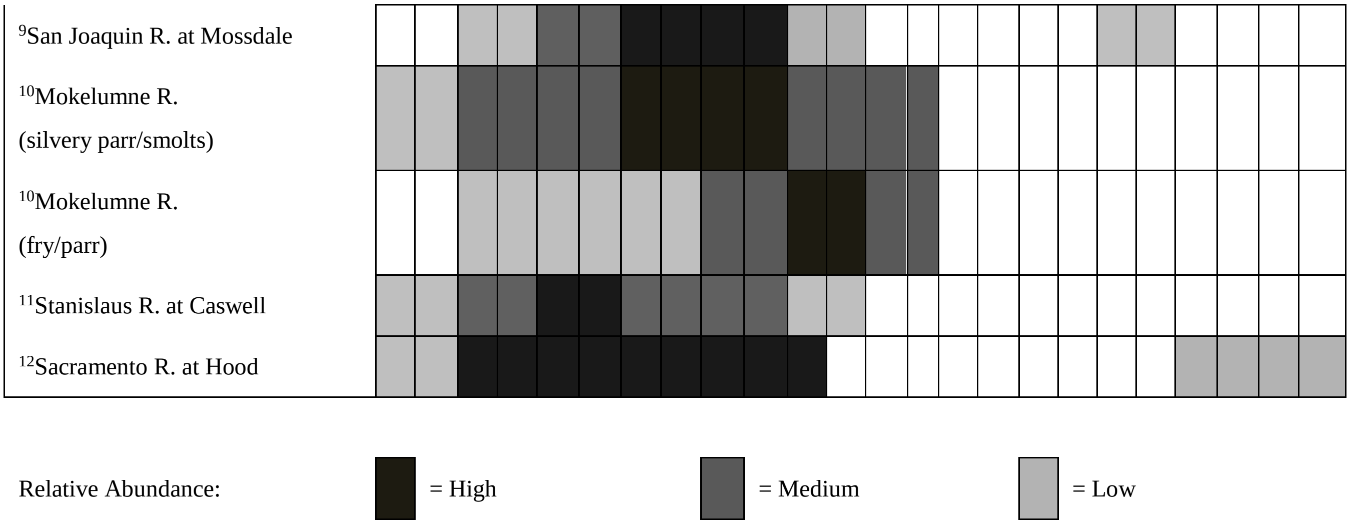
1.3.3.5 Kelts

Post-spawning steelhead (kelts) may migrate downstream to the ocean immediately after spawning, or they may spend several weeks holding in pools before outmigrating (Shapovalov and Taft 1954). Recent studies have shown that kelts may remain in freshwater for an entire year after spawning (Teo *et al.* 2011), but that most return to the ocean (Null 2013).

Table 1-4 shows the temporal occurrence of (a) adult and (b) juvenile California Central Valley steelhead at locations in the Central Valley. Darker shades indicate months of greatest relative abundance.

Table 1-4. The temporal occurrence of (a) adult and (b) juvenile California Central Valley steelhead at locations in the Central Valley.

(a) Adult migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[1]Sacramento R. at Fremont Weir												
[2]Sacramento R. at RBDD												
[3]Mill & Deer Creeks												
[4]Mill Creek at Clough Dam												
[5]San Joaquin River												
(b) Juvenile migration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[1,2]Sacramento R. near Fremont Weir												
[6]Sacramento R. at Knights Landing												
[7]Mill & Deer Creeks (silvery parr/smolts)												
[7]Mill & Deer Creeks (fry/parr)												
[8]Chipps Island (clipped)												
[8]ChippsIsland (unclipped)												



Sources: [1](Hallock 1957); [2](McEwan 2001); [3](Harvey 1995); [4]CDFW unpublished data; [5]CDFG Steelhead Report Card Data 2007; [6]NMFS analysis of 1998-2011 CDFW data; [7](Johnson and Merrick 2012); [8]NMFS analysis of 1998-2011 USFWS data; [9]NMFS analysis of 2003-2011 USFWS data; [10]unpublished EBMUD RST data for 2008-2013; [11]Oakdale RST data (collected by FishBio) summarized by John Hannon (Reclamation) ; [12](Schaffter 1980).

1.3.4 Description of Viable Salmonid Population (VSP) Parameters

As an approach to determining the conservation status of salmonids, NMFS has developed a framework for identifying attributes of a viable salmonid population (VSP). The intent of this framework is to provide parties with the ability to assess the effects of management and conservation actions and ensure their actions promote the listed species' survival and recovery. This framework is known as the VSP concept (McElhany *et al.* 2000b). The VSP concept measures population performance in terms of four key parameters: abundance, population growth rate, spatial structure, and diversity.

1.3.4.1 Abundance

Historic CCV steelhead run sizes are difficult to estimate given the paucity of data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s the CCV steelhead run size had declined to about 40,000 adults (McEwan 2001). Hallock *et al.* (1961) estimated an average of 20,540 adult steelhead through the 1960s in the Sacramento River upstream of the Feather River. Steelhead counts at the Red Bluff Diversion Dam (RBDD) declined from an average of 11,187 from 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system, based on RBDD counts, to be no more than 10,000 adults (McEwan and Jackson 1996, McEwan 2001). Steelhead escapement surveys at RBDD ended in 1993 due to changes in dam operations. Comprehensive steelhead population monitoring has not taken place in the Central Valley since then, despite 100 percent marking of hatchery steelhead smolts since 1998. Efforts are underway to improve this deficiency, and a long-term adult escapement monitoring plan is being formulated (Eilers *et al.* 2010).

Current abundance data is limited to returns to hatcheries and redd surveys conducted on a few rivers. The hatchery data is the most reliable, as redd surveys for steelhead are often made difficult by high flows and turbid water usually present during the winter-spring spawning period.

Coleman National Fish Hatchery (NFH) operates a weir on Battle Creek, where all upstream fish movement is blocked August through February, during the hatchery spawning season. Counts of steelhead captured at and passed above this weir represent one of the better data sources for the CCV DPS. However, changes in hatchery policies and transfer of fish complicate the interpretation of these data. In 2005, per NMFS request, Coleman NFH stopped transferring all adipose-fin clipped steelhead above the weir, resulting in a large decrease in the overall numbers of steelhead above the weir in recent years. In addition, in 2003, Coleman NFH transferred about 1,000 clipped adult steelhead to Keswick Reservoir, and these fish are not included in the data. The result is that the only unbiased time series for Battle Creek is the number of unclipped (wild) steelhead since 2001, which have declined slightly since that time, mostly because of the high returns observed in 2002 and 2003.

Prior to 2002, hatchery and natural-origin steelhead in Battle Creek were not differentiable, and all steelhead were managed as a single, homogeneous stock, although USFWS believes the majority of returning fish in years prior to 2002 were hatchery-origin. Abundance estimates of natural-origin steelhead in Battle Creek began in 2001. These estimates of steelhead abundance include all *O. mykiss*, including resident and anadromous fish (Figure 1-8).

Steelhead returns to Coleman NFH have increased during the last few years, 2011 to 2014 (Figure 1-8). After hitting a low of only 790 fish in 2010, 2013 and 2014 have averaged 2,895 fish (Figure 1-8). Since 2003, adults returning to the hatchery have been classified as wild (unclipped) or hatchery produced (adipose fin clipped). Wild adults counted at the hatchery each year represent a small fraction of overall returns, but their numbers have remained relative steady, typically 200-300 fish each year. Numbers of wild adults returning each year have ranged from 252 to 610 from 2010 to 2014 (Figure 1-8).

Redd counts are conducted in the American River and in Clear Creek (Shasta County). An average of 143 redds have been counted on the American River from 2002-2015 [(Figure 1-9; data from (Hannon *et al.* 2003, Hannon and Deason 2008, Chase 2010)]. Surveys were not conducted in some years on the American River due to high flows and low visibility. An average of 178 redds have been counted in Clear Creek from 2001 to 2015 (Figure 1-10; data from USFWS). The Clear Creek steelhead population appears to have increased in abundance since Saeltzer Dam was removed in 2000, as the number of redds observed in surveys conducted by the USFWS has steadily increased since 2001 (Figure 1-10). The average redd index from 2001 to 2011 is 178, representing a range of approximately 100-1023 spawning adult steelhead on average each year, based on an approximate observed adult-to-redd ratio in Clear Creek (U.S. Fish and Wildlife Service 2015). The vast majority of these steelhead are wild fish, as no hatchery steelhead are stocked in Clear Creek.

The East Bay Municipal Utilities District (EBMUD) has included steelhead in their redd surveys on the Lower Mokelumne River since the 1999-2000 spawning season, and the overall trend is a slight increase. However, it is generally believed that most of the *O. mykiss* spawning in the Mokelumne River are resident fish (Satterthwaite *et al.* 2010), which are not part of the CCV steelhead DPS. In the most recent 5-year status review, NMFS did not to include the Mokelumne River steelhead population in the DPS (National Marine Fisheries Service 2016c).

The returns of CCV steelhead to the Feather River Hatchery experienced a sharp decrease from 2003 to 2010, with only 679, 312, and 86 fish returning in 2008, 2009 and 2010, respectively (Figure 1-11). In recent years, however, returns have experienced an increase with 830, 1797, and 1505 fish returning in 2012, 2013 and 2014, respectively. Almost all these fish are hatchery fish, and stocking levels have remained fairly constant, suggesting that smolt and/or ocean survival was poor for age classes that showed poor returns in the late 2000s.

Catches of steelhead at the fish collection facilities in the southern Delta are another source of information on the relative abundance of the CCV steelhead DPS, as well as the proportion of wild steelhead relative to hatchery steelhead (CDFG; ftp.delta.dfg.ca.gov/salvage). The overall catch of steelhead at these facilities has been highly variable since 1993 (Figure 1-13). Variability in catch is likely due to differences in water year types as Delta exports fluctuate. The percentage of unclipped steelhead in salvage has also fluctuated, but has generally declined since 100 percent clipping started in 1998. The number of stocked hatchery steelhead has remained relatively constant overall since 1998, even though the number stocked in any individual hatchery has fluctuated.

The years 2009 and 2010 showed poor returns of steelhead to the Feather River Hatchery and Coleman Hatchery, probably due to three consecutive drought years in 2007-2009, which would have impacted parr and smolt growth and survival in the rivers, and possibly due to poor coastal upwelling conditions in 2005 and 2006, which strongly impacted fall-run Chinook salmon post-smolt survival (Lindley *et al.* 2009b). Wild (unclipped) adult counts appear not to have decreased as greatly in those same years, based on returns to the hatcheries and redd counts conducted on Clear Creek, and the American and Mokelumne rivers. This may reflect greater fitness of naturally produced steelhead relative to hatchery fish, and certainly merits further study.

Overall, steelhead returns to hatcheries have fluctuated so much from 2001 to 2015 that no clear trend is present, other than the fact that the numbers are still far below those seen in the 1960s and 1970s, and only a tiny fraction of the historical estimate. Returns of natural origin fish are very poorly monitored, but the little data available suggest that the numbers are very small, though perhaps not as variable from year to year as the hatchery returns.

Figure 1-8 depicts steelhead returns to Coleman NFH from 1988-2014. Starting in 2001, fish were classified as either wild (unclipped) or hatchery produced (clipped).

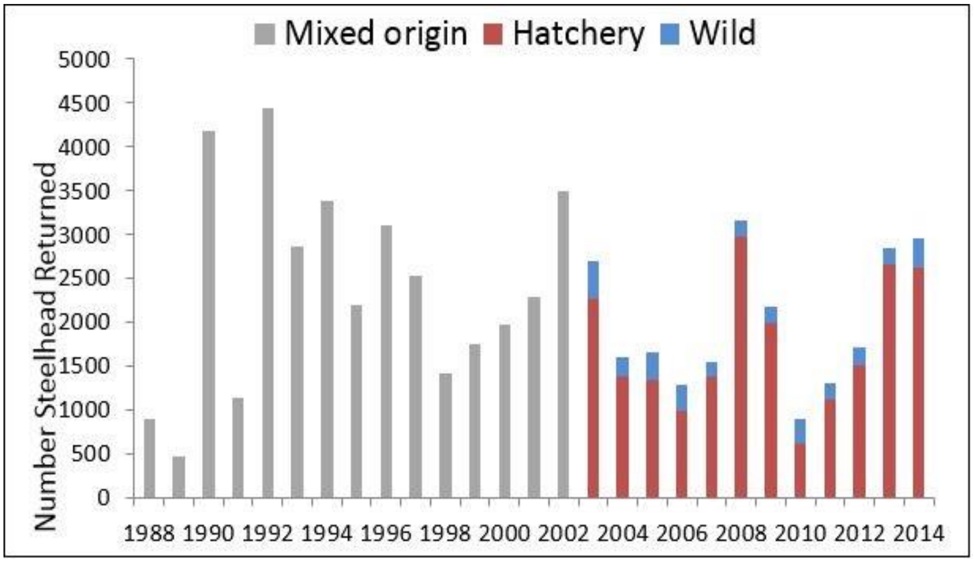


Figure 1-8. Steelhead returns to Coleman NFH from 1988-2014.

Figure 1-9 shows steelhead redd counts from surveys on the American River from 2002-2015. Surveys could not be conducted in some years due to high flows and low visibility.

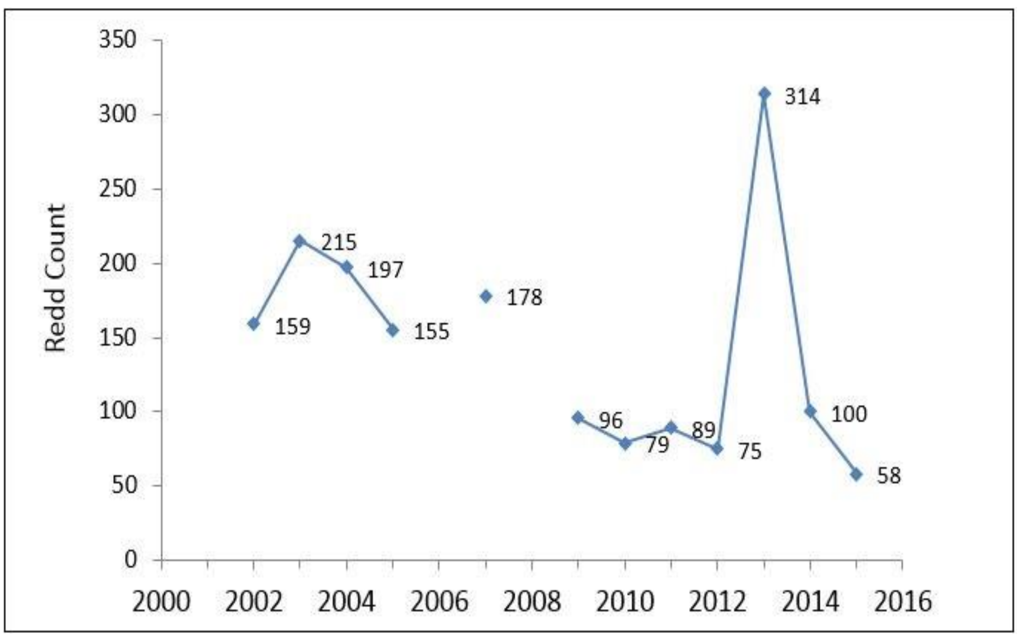


Figure 1-9. Steelhead redd counts from surveys on the American River from 2002-2015.

Figure 1-10 shows redd counts from USFWS surveys on Clear Creek from 2001-2015.

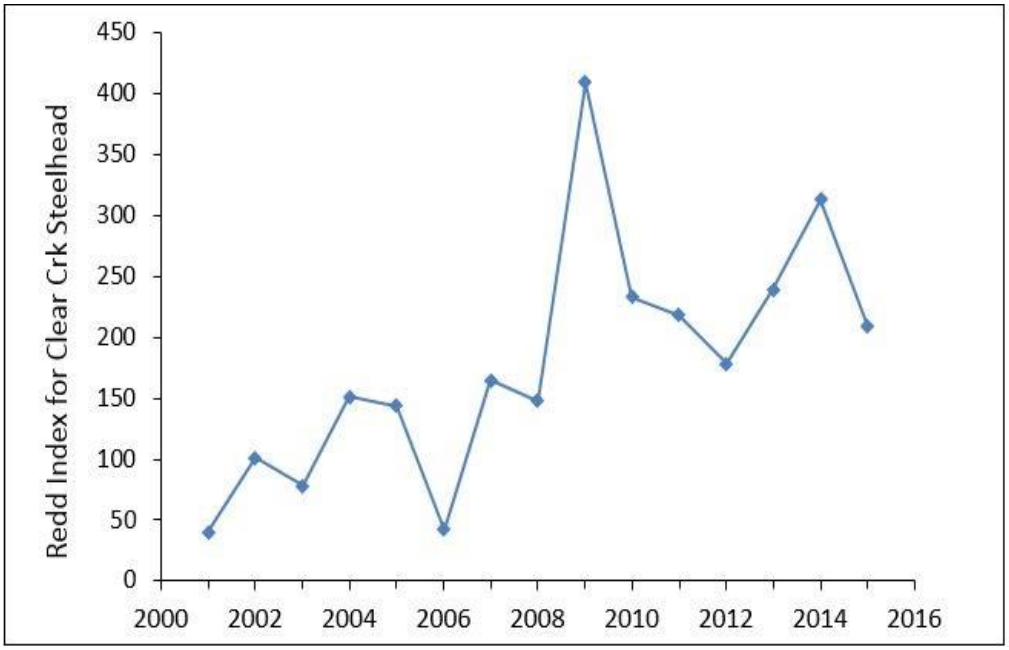


Figure 1-10. Redd counts from USFWS surveys on Clear Creek from 2001-2015.

Figure 1-11 shows steelhead returns to the Feather River Hatchery from 1964-2015.

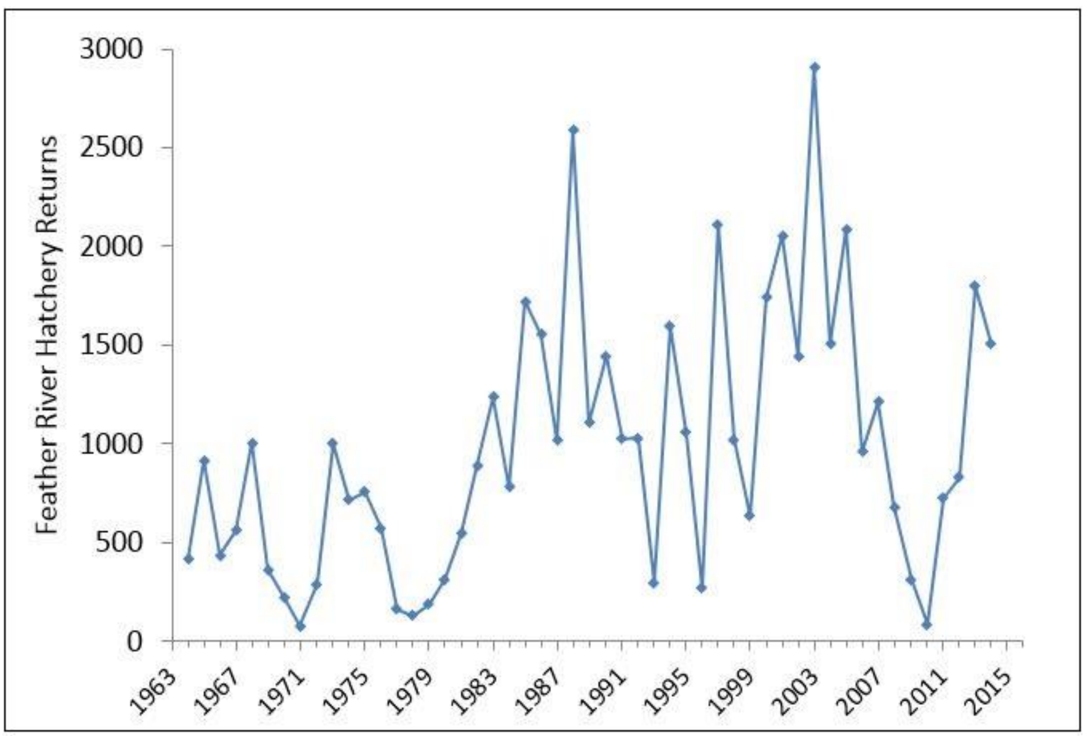


Figure 1-11. Steelhead returns to the Feather River Hatchery from 1964-2015.

1.3.4.2 Productivity

An estimated 100,000 to 300,000 naturally produced juvenile steelhead are estimated to leave the Central Valley annually, based on rough calculations from sporadic catches in trawl gear (Good *et al.* 2005a). The Mossdale trawls on the San Joaquin River conducted annually by CDFW and USFWS capture steelhead smolts, although usually in very small numbers. These steelhead recoveries, which represent migrants from the Stanislaus, Tuolumne, and Merced rivers, suggest that the productivity of CCV steelhead in these tributaries is very low. Also, the Chipps Island midwater trawl dataset from the USFWS provides information on the trend (Williams *et al.* 2011).

Nobriga and Cadrett (2001) used the ratio of adipose fin-clipped (hatchery) to unclipped (wild) steelhead smolt catch ratios in the Chipps Island trawl from 1998 through 2000 to estimate that about 400,000 to 700,000 steelhead smolts are produced naturally each year in the Central Valley. Good *et al.* (2005a) made the following conclusion based on the Chipps Island data.

> *If we make the fairly generous assumptions (in the sense of generating large estimates of spawners) that average fecundity is 5,000 eggs per female, 1 percent of eggs survive to reach Chipps Island, and 181,000 smolts are produced (the 1998-2000 average), about 3,628 female steelhead spawn naturally in the entire Central Valley. This can be compared with McEwan (2001) estimate of 1 million to 2 million spawners before 1850, and 40,000 spawners in the 1960s.*

The Chipps Island midwater trawl dataset maintained by the USFWS provides information on the trend in abundance for the CCV steelhead DPS as a whole. Updated through 2014, the trawl data indicate that the level of natural production of steelhead has remained very low since the 2011 status review (Figure 1-12). Catch per unit effort (CPUE) has fluctuated but remained relatively constant over the past decade, but the proportion of the catch that is adipose-clipped (100% of hatchery steelhead production have been adipose fin-clipped starting in 1998) has risen, exceeding 90 percent in some years and reaching a high of 95 percent in 2010 (Williams *et al.* 2011). Because hatchery releases have been fairly constant, this implies that natural production of juvenile steelhead has been declining in the Central Valley.

The top of Figure 1-12 shows the catch of steelhead at Chipps Island by the USFWS midwater trawl survey. The middle section shows the fraction of the catch bearing an adipose fin clip. 100 percent of steelhead production has been marked starting in 1998, denoted with the vertical gray line. The bottom section shows CPUE in fish per million m-3 swept volume. CPUE is not easily comparable across the entire period of record, as over time, sampling has occurred over more of the year and catches of juvenile steelhead are expected to be low outside of the primary migratory season.

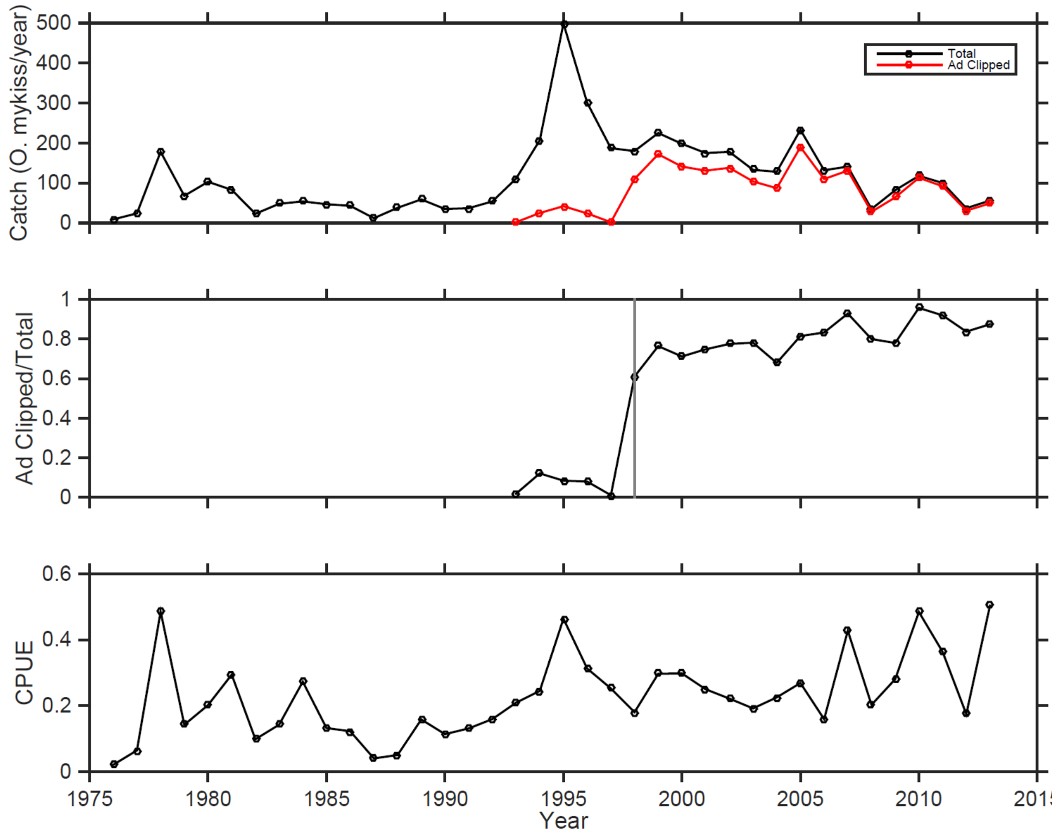


Figure 1-12. Steelhead Catch at Chipps Island midwater trawl (USFWS unpublished data)

In the Mokelumne River, East Bay Municipal Utilities District (EBMUD) has included steelhead in their redd surveys on the Lower Mokelumne River since the 1999-2000 spawning season (National Marine Fisheries Service 2011b). Based on data from these surveys, the overall trend suggests that redd numbers have slightly increased over the years (2000-2010). However, according to Satterthwaite *et al.* (2010), it is likely that most of the *O. mykiss* spawning in the Mokelumne River are non-anadromous (or resident) fish rather than steelhead. The Mokelumne River steelhead population is supplemented by Mokelumne River Hatchery production. In the past, this hatchery received fish imported from the Feather River and Nimbus hatcheries (Merz 2002). This practice was discontinued, however, for Nimbus stock after 1991 and discontinued for Feather River stock after 2008. Genetic studies show that the Mokelumne River Hatchery steelhead are closely related to Feather River fish, suggesting that there has been little carry-over of genes from the Nimbus stock (Garza and Pearse 2015).

Additionally, on the Mokelumne River, it appears that many fish can reach a size large enough to smolt at age 1, but the slower-growing fish are better served to mature as young-of-year (YOY) and spawn at age 1 rather than risk the extra freshwater mortality associated with waiting to smolt at age 2 (because much less time must elapse before the age 1 spawning opportunity compared to age 2 emigration). Once the first spawning opportunity has passed, however, and even slow growing fish are large enough to have a moderate chance of survival in the ocean, it takes too long and exposes fish to too much risk of freshwater mortality to grow to a large enough size to spawn with much success as a resident female at an even older age (Satterthwaite *et al.* (2010).

These results suggest that restoration activities for CCV steelhead should focus on habitat improvements that both increase parr survival and growth in natal rivers, especially in the summer and fall, and improve smolt survival in the lower river reaches, the Delta, and Bays.

Catches of steelhead at the fish collection facilities in the southern Delta are another source of information on the relative abundance of the CCV steelhead DPS, as well as the production of wild steelhead relative to hatchery steelhead (ftp.delta.dfg.ca.gov/salvage). The overall catch of steelhead has declined dramatically since the early 2000s, with an overall average of 2,705 in the last 10 years (2004 to 2014), as measured by expanded salvage (Figure 1-13). The percentage of wild (unclipped) fish in salvage has fluctuated, but has leveled off to an average of 36 percent since a high of 93 percent in 1999. The number of stocked hatchery steelhead has remained relatively constant overall since 1998, even though the number stocked in any individual hatchery has fluctuated. This relatively constant hatchery production, coupled with the dramatic decline in hatchery-origin steelhead catch at the south Delta fish collection facilities suggests that either stocked hatchery fish from the Sacramento basin are using a more natural outmigration path and are not being pulled into the south Delta fish facilities or the immediate survival of those stocked fish has decreased. With respect to wild steelhead, the data shown in Figure 1-12 indicate that over the last few years (2011 to 2014) fewer adults are spawning (fewer eggs deposited), survival of early life stages has decreased, and/or wild steelhead are experiencing reduced exposure to the south Delta fish facilities.

Figure 1-13 depicts steelhead salvaged in the Delta fish collection facilities from 1993 to 2014. All hatchery steelhead have been adipose fin-clipped since 1998. Data are from CDFW, at: ftp.delta.dfg.ca.gov/salvage.

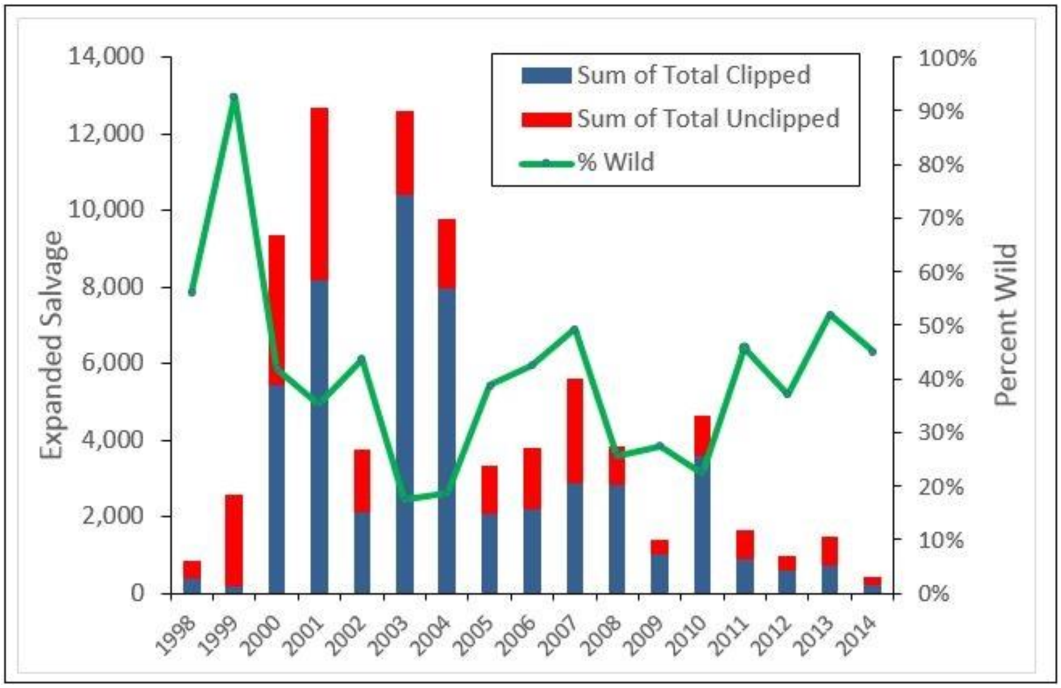


Figure 1-13. Steelhead salvaged in the Delta fish collection facilities.

Since 2003, fish returning to the Coleman National Fish Hatchery have been identified as wild (adipose fin intact) or hatchery produced (ad-clipped). Returns of wild fish to the hatchery have remained fairly steady at 200-300 fish per year, but represent a small fraction of the overall hatchery returns. Numbers of hatchery origin fish returning to the hatchery have fluctuated much more widely; ranging from 624 to 2,968 fish per year (Figure 1-8).

1.3.4.3 Spatial Structure

About 80 percent of the historical spawning and rearing habitat once used by anadromous *O. mykiss* in the Central Valley is now upstream of impassible dams (Lindley *et al.* 2006). The extent of habitat loss for steelhead most likely was much higher than that for salmon because steelhead were undoubtedly more extensively distributed. Due to their superior jumping ability, the timing of their upstream migration, which coincided with the winter rainy season, and their less restrictive preferences for spawning gravels, steelhead could have utilized at least hundreds of miles of smaller tributaries not accessible to the earlier-spawning salmon (Yoshiyama *et al.* 1996). Many historical populations of CCV steelhead are entirely above impassable barriers and may persist as resident or adfluvial rainbow trout, although they are presently not considered part of the DPS. Steelhead were found as far south as the Kings River (and possibly Kern River systems in wet years) (McEwan 2001). Native American groups such as the Chunut people have had accounts of steelhead in the Tulare Basin (Latta 1977).

Steelhead are well-distributed throughout the Central Valley below the major rim dams (Good *et al.* 2005a, National Marine Fisheries Service 2016c). Zimmerman *et al.* (2009) used otolith microchemistry to show that *O. mykiss* of anadromous parentage occur in all three major San Joaquin River tributaries, but at low levels, and that these tributaries have a higher percentage of resident *O. mykiss* compared to the Sacramento River and its tributaries.

Monitoring has detected small numbers of steelhead in the Stanislaus, Mokelumne, and Calaveras rivers, and other streams previously thought to be devoid of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (S.P. Cramer & Associates 2000). A counting weir has been in place in the Stanislaus River since 2002 and in the Tuolumne River since 2009 to detect adult salmon; these weirs have also detected *O. mykiss* passage. In 2012, 15 adult *O. mykiss* were detected passing the Tuolumne River weir and 82 adult *O. mykiss* were detected at the Stanislaus River weir (FISHBIO LLC 2012, FISHBIO 2013a). Also, rotary screw trap sampling has occurred since 1995 in the Tuolumne River, but only one juvenile *O. mykiss* was caught during the 2012 season (FISHBIO 2013b). Rotary screw traps are well known to be very inefficient at catching steelhead smolts, so the actual numbers of smolts produced in these rivers could be much higher. Rotary screw trapping on the Merced River has occurred since 1999. A fish counting weir was installed on this river in 2012. Since installation, one adult *O. mykiss* has been reported passing the weir. Juvenile *O. mykiss* were not reported captured in the rotary screw traps on the Merced River until 2012, when a total of 381 were caught (FISHBIO LLC 2013). The unusually high number of *O. mykiss* captured may be attributed to a flashy storm event that rapidly increased flows over a 24-hour period. Annual Kodiak trawl surveys are conducted on the San Joaquin River at Mossdale by CDFW. A total of 17 *O. mykiss* were caught during the 2012 season (California Department of Fish and Wildlife 2013).

Most of the steelhead populations in the Central Valley have a high hatchery component, including Battle Creek (adults intercepted at the Coleman NFH weir), the American River, Feather River, and Mokelumne River. This is confounded, of course, by the fact that most of the dedicated monitoring programs in the Central Valley occur on rivers that are annually stocked. Clear Creek and Mill Creek are the exceptions.

Implementation of CDFW's Steelhead Monitoring Program began during the fall of 2015. Important components of the program include a Mainstem Sacramento River Steelhead Mark-Recapture Program and an Upper Sacramento River Basin Adult Steelhead Video/DIDSON Monitoring Program. The monitoring program will use a temporally stratified mark-recapture survey design in the lower Sacramento River, employing wire fyke traps to capture, mark, and recapture upstream migrating adult steelhead to estimate adult steelhead escapement from the Sacramento-San Joaquin River Delta. Data collected from recaptured adult steelhead will provide additional information on tributary escapement, survival, population structure, population distribution, and spatial and temporal behavior of both hatchery- and natural-origin steelhead.

The low adult returns to the San Joaquin tributaries and the low numbers of juvenile emigrants typically captured suggest that existing populations of CCV steelhead on the Tuolumne, Merced, and lower San Joaquin rivers are severely depressed. The loss of these populations would severely impact CCV steelhead spatial structure and further challenge the viability of the CCV steelhead DPS.

Efforts to provide passage of salmonids over impassable dams have the potential to increase the spatial diversity of Central Valley steelhead populations if the passage programs are implemented for steelhead. In addition, the SJRRP calls for a combination of channel and structural modifications along the San Joaquin River below Friant Dam, releases of water from Friant Dam to the confluence of the Merced River, and the reintroduction of spring-run and fall-run Chinook salmon. If the SJRRP is successful, habitat improved for spring-run Chinook salmon could also benefit CCV steelhead (National Marine Fisheries Service 2016c).

1.3.4.4 Diversity

1.3.4.4.1 Genetic Diversity

California Central Valley steelhead abundance and growth rates continue to decline, largely the result of a significant reduction in the amount and diversity of habitats available to these populations (Lindley *et al.* 2006). Recent reductions in population size are also supported by genetic analysis (Nielsen *et al.* 2003).

Garza and Pearse (2008) analyzed the genetic relationships among CCV steelhead populations and found that unlike the situation in coastal California watersheds, fish below barriers in the Central Valley were often more closely related to below barrier fish from other watersheds than to *O. mykiss* above barriers in the same watershed. This pattern suggests the ancestral genetic structure is still relatively intact above barriers, but may have been altered below barriers by stock transfers.

The genetic diversity of CCV steelhead is also compromised by hatchery origin fish, which likely comprise the majority of the annual spawning runs, placing the natural population at a high risk of extinction (Lindley *et al.* 2007a). There are four hatcheries (Coleman National Fish Hatchery, Feather River Fish Hatchery, Nimbus Fish Hatchery, and Mokelumne River Fish Hatchery) in the Central Valley which combined release approximately 1.6 million yearling steelhead smolts each year. These programs are intended to mitigate for the loss of steelhead habitat caused by dam construction, but hatchery origin fish now appear to constitute a major proportion of the total abundance in the DPS. Two of these hatchery stocks (Nimbus and Mokelumne River hatcheries) originated from outside the DPS (primarily from the Eel and Mad rivers) and are not presently considered part of the DPS. However, during the recent NMFS five-year status review for CV steelhead, NMFS recommended including the Mokelumne River hatchery steelhead population in the CV Steelhead DPS due to the close genetic relationship with Feather River hatchery steelhead that are considered part of the native Central Valley stock (National Marine Fisheries Service 2016b).

1.3.4.4.2 Life-History Diversity

Steelhead in the Central Valley historically consisted of both summer-run and winter-run migratory forms, based on their state of sexual maturity at the time of river entry and the duration of their time in freshwater before spawning.

> *Between 1944 and 1947, annual counts of summer-run steelhead passing through the Old Folsom Dam fish ladder during May, June, and July ranged from 400 to 1,246 fish. After 1950, when the fish ladder at Old Folsom Dam was destroyed by flood flows, summer-run steelhead were no longer able to access their historic spawning areas, and perished in the warm water downstream of Old Folsom Dam (Gerstung 1971).*

Only winter-run (ocean maturing) steelhead currently are found in California Central Valley rivers and streams (McEwan and Jackson 1996, Moyle 2002a). Summer-run steelhead have been extirpated due to a lack of suitable holding and staging habitat, such as cold-water pools in the headwaters of CV streams, presently located above impassible dams (Lindley *et al.* 2006).

Juvenile steelhead (parr) rear in freshwater for one to three years before migrating to the ocean as smolts (Moyle 2002a). The time that parr spend in freshwater is inversely related to their growth rate, with faster-growing members of a cohort smolting at an earlier age but a smaller size (Seelbach 1993, Peven *et al.* 1994). Hallock *et al.* (1961) aged 100 adult steelhead caught in the Sacramento River upstream of the Feather River confluence in 1954 and found that 70 had smolted at age-2, 29 at age-1, and one at age-3. Seventeen of the adults were repeat spawners, with three fish on their third spawning migration, and one on its fifth. Age at first maturity varies among populations. In the Central Valley, most steelhead return to their natal streams as adults at a total age of two to four years (Hallock *et al.* 1961, McEwan and Jackson 1996).

Deer and Mill creeks were monitored from 1994 to 2010 by the CDFW using rotary screw traps to capture emigrating juvenile steelhead (Johnson and Merrick 2012). Fish in the fry stage averaged 34 and 41 mm FL in Deer and Mill, respectively, while those in the parr stage averaged 115 mm FL in both streams. Silvery parr averaged 180 and 181 mm in Deer and Mill creeks, while smolts averaged 210 and 204 mm. Most silvery parr and smolts were caught in the spring months from March through May, while fry and parr peaked later in the spring (May and June) and were fairly common in the fall (October through December) as well.

In contrast to the upper Sacramento River tributaries, Lower American River juvenile steelhead have been shown to smolt at a very large size (270 to 350 mm FL), and nearly all smolt at age-1 (Sogard *et al.* 2012).

1.3.4.5 Summary of DPS Viability

All indications are that natural CCV steelhead have continued to decrease in abundance and in the proportion of natural fish over the past 25 years (Good *et al.* 2005a, National Marine Fisheries Service 2016c); the long-term trend remains negative. Hatchery production and returns are dominant over natural fish, and one of the four hatcheries is dominated by Eel/Mad River origin steelhead stock.

The ratio between naturally produced juvenile steelhead to hatchery juvenile steelhead in fish monitoring efforts indicates that the wild population abundance has remained at a relatively steady state since the 2011 status review and remains much lower than percentages observed in previous decades. Hatchery releases (100 percent adipose fin-clipped fish since 1998) have remained relatively constant over the past decade, yet the proportion of adipose fin-clipped hatchery smolts to unclipped naturally produced smolts has steadily increased over the past decade.

Although there have been recent restoration efforts in the San Joaquin River tributaries, CCV steelhead populations in the San Joaquin Basin continue to show an overall very low abundance, and fluctuating return rates. Lindley *et al.* (2007a) developed viability criteria for Central Valley salmonids. Using data through 2005, Lindley *et al.* (2007a) found that data were insufficient to determine the status of any of the naturally-spawning populations of CCV steelhead, except for those spawning in rivers adjacent to hatcheries, which were likely to be at high risk of extinction due to extensive spawning of hatchery-origin fish in natural areas.

The widespread distribution of wild steelhead in the Central Valley provides the spatial structure necessary for the DPS to survive and avoid localized catastrophes. However, most wild CCV populations are very small and may lack the resiliency to persist for protracted periods if subjected to additional stressors, particularly widespread stressors such as climate change. The genetic diversity of CCV steelhead has likely been impacted by low population sizes and high numbers of hatchery fish relative to wild fish. The life-history diversity of the DPS is mostly unknown because very few studies have been published on traits such as age structure, size at age, or growth rates in CCV steelhead.

The most recent status review of the CCV steelhead DPS (National Marine Fisheries Service 2016c) found that the status of the DPS appears to have remained unchanged since the 2011 status review (Good *et al.* 2005a), and the DPS is likely to become endangered within the foreseeable future throughout all or a significant portion of its range.

1.4 Southern Distinct Population Segment (sDPS) of North American Green Sturgeon (*Acipenser medirostris*)

- Listed as threatened (April 7, 2006, 71 FR 17757).
- Critical habitat designated (October 9, 2009, 74 FR 52300).

1.4.1 Species Listing and Critical Habitat Designation History

Two distinct population segments (DPS) of North American green sturgeon have been identified; a northern DPS (nDPS) and a southern DPS (sDPS). While individuals from the two DPSs are visually indistinguishable and have significant geographical overlap, current information indicates that they do not interbreed or utilize the same natal streams (68 FR 4433, January 23, 2003; Adams et al. 2002; Isreal et al. 2004). This section discusses the sDPS green sturgeon, which is listed under the ESA, and its designated critical habitat. The sDPS green sturgeon consists of green sturgeon originating from the Sacramento River basin and from coastal rivers south of the Eel River (April 7, 2006, 71 FR 17757). When necessary to fill in knowledge gaps, we use available life history information for white sturgeon (*A. transmontanus)* and other sturgeon species, noting the use of other species life history information as a surrogate.

In June of 2001, NMFS received a petition to list green sturgeon and designate their critical habitat under the ESA. After completion of a status review (Adams *et al.* 2002), NMFS found that the species was comprised of two DPSs that qualify as species under the ESA, but that neither DPS warranted listing (January 23, 2003, 68 FR 4433). Several entities challenged our determination that listing was not warranted in federal district court, and the court issued an order setting aside and remanding our determination. Following a status review update in 2005, NMFS listed the sDPS as threatened based on the reduction of potential spawning habitat, the severe threats to the single remaining spawning population (in the Sacramento River), the inability to alleviate these threats with the conservation measures in place, and the decrease in observed numbers of juvenile green sturgeon collected in the past two decades before listing compared to those collected historically (April 7, 2006, 71 FR 17757). Since the 2006 listing decision, new information has become available regarding the many threats to the species from entrainment, flow operations, reservoir operations, habitat loss, water quality, toxics, invasive species and population dynamics; reaffirming NMFS concerns that sDPS green sturgeon face substantial threats to their viability and recovery (Israel and Klimley 2008b).

1.4.2 Critical Habitat Physical and Biological Features for sDPS green sturgeon

Critical habitat for sDPS green sturgeon includes:

(1) The Sacramento River from the Sacramento I-Street Bridge to Keswick Dam, including the Sutter and Yolo Bypasses and the lower American River from the confluence with the mainstem Sacramento River upstream to the highway 160 bridge,

(2) The Feather River from its confluence with the Sacramento River upstream to the Fish Barrier Dam,

(3) The Yuba River from the confluence with the Feather River upstream to Daguerre Point Dam,

(4) The Sacramento-San Joaquin Delta (as defined by California Water Code section 12220, except for listed excluded areas),

(5) San Francisco, San Pablo, Suisun and Humboldt bays in California,

(6) Coos, Winchester, Yaquina, and Nehalem bays in Oregon,

(7) Willapa Bay and Grays Harbor in Washington,

(8) the lower Columbia River estuary from the mouth to river kilometer 74, and

(9) all U.S. coastal marine waters out to the 60-fathom depth bathymetry line, from Monterey Bay, California north and east to include the Strait of Juan de Fuca, Washington (October 9, 2009, 74 FR 52300) (Figure 1-14).

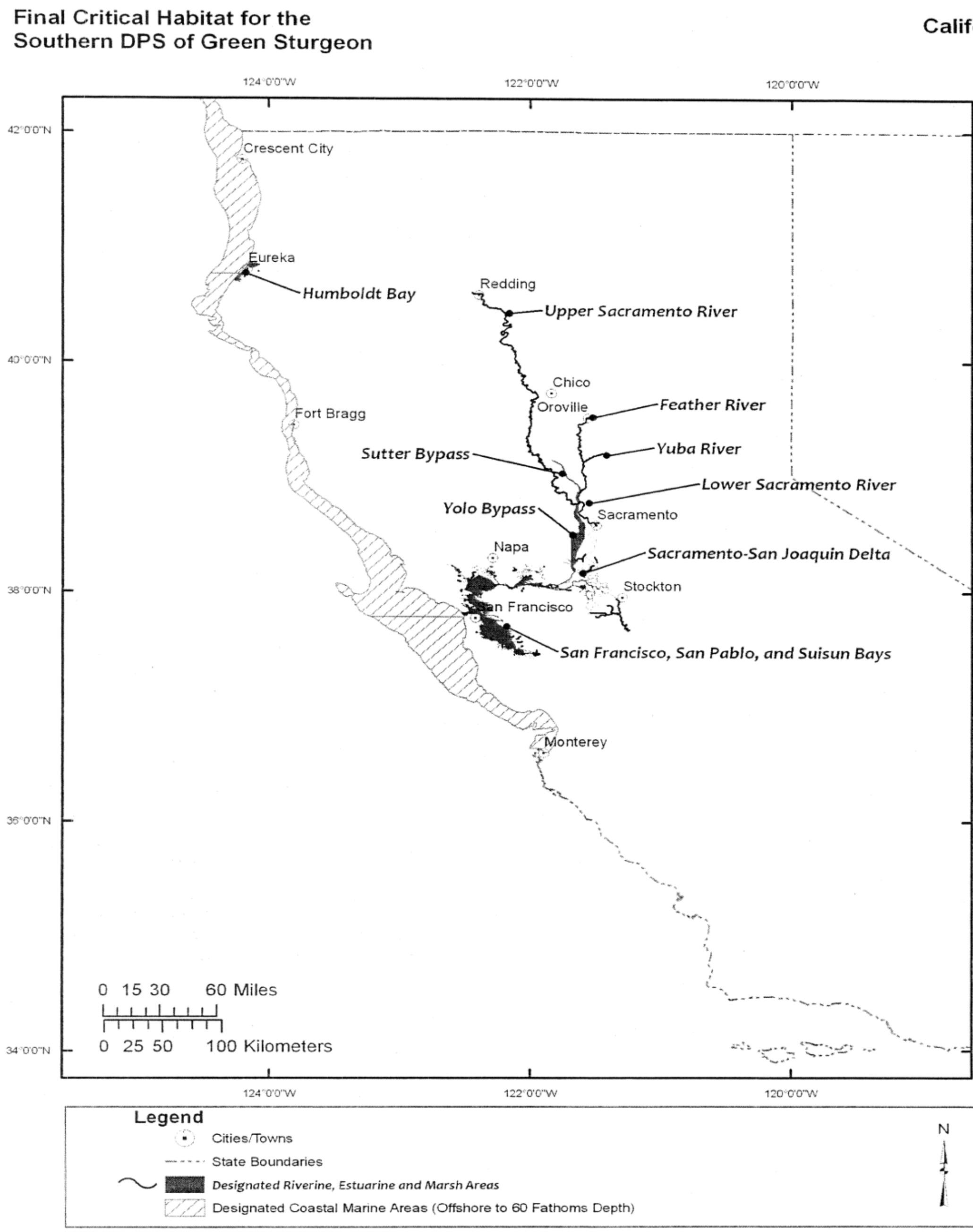


Figure 1-14. Green sturgeon critical habitat in California. Source: October 9, 2009, 74 FR 52300

The following subsections describe the status of the PBFs of sDPS green sturgeon critical habitat, which are listed in the critical habitat designation (October 9, 2009, 74 FR 52300). The specific PBFs in freshwater riverine systems include the following.

1.4.2.1 Food Resources

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include food resources – abundant prey items for larval, juvenile, subadult, and adult life stages. Green sturgeon food resources likely include drifting and benthic invertebrates, forage fish, and fish eggs. In a stomach content analysis, Radtke (1966a) found that the diet of juvenile green sturgeon consisted primarily of mysid shrimp (*Neomysis awatschensis*) and amphipods (*Corophium*). Although little specific information on food resources is available for green sturgeon at various lifecycle stages within freshwater riverine systems, they are presumed to be opportunistic feeders with a diet similar to other sturgeon such as white sturgeon, which also occupy the Sacramento River basin (Israel and Klimley 2008a). Seasonally abundant drifting and benthic invertebrates have been shown to be the major food items for white sturgeon in the lower Columbia River (Muir *et al.* 2000). Increasing size of prey items in white sturgeon has also been positively correlated with increasing sizes of individual fish (Muir *et al.* 2000). The establishment of non-native species of plants and invertebrates (e.g., mussels, clams), which is occurring in the Sacramento-San Joaquin River Delta, has the potential to alter food resources for the sDPS and those effects could be exacerbated by climate change (Draft GSRP 2016). Research conducted on white sturgeon and to a lesser extent, green sturgeon, has shown that many of their non-native food resources including the overbite clam, *Corbula amurensis*, has become a common food source for sturgeon and is either non-digestible (Kogut 2008) or, if digested, may be exposing green sturgeon to high levels of selenium (CDFG 2002; Linville et al. 2002). Bioaccumulation of selenium has known impacts on fish viability and reproduction.

The PBFs of sDPS green sturgeon critical habitat in estuarine habitats include food resources - abundant prey items within estuarine habitats and substrates for juvenile, subadult, and adult life stages. Prey species for juvenile, subadult, and adult green sturgeon within bays and estuaries primarily consist of benthic invertebrates and fish, including crangonid shrimp, callianassid shrimp, burrowing thalassinidean shrimp, amphipods, isopods, clams, annelid worms, crabs, sand lances, and anchovies. These prey species are critical for rearing, foraging, growth, and development of juvenile, subadult, and adult green sturgeon within bays and estuaries. As discussed above, non-native species are impacting the prey availability for sDPS in estuarine areas. The extent and severity of this impact is unknown.

The PBFs of sDPS green sturgeon critical habitat in nearshore coastal marine areas include food resources - abundant prey items for subadults and adults, which may include benthic invertebrates and fishes. Little is known about the prey base of sDPS in these areas (Draft GSRP 2016).

1.4.2.2 Substrate Type or Size

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include substrate type or size (i.e., structural features of substrates) - substrates suitable for egg deposition and development (*e.g.*, bedrock sills and shelves, cobble and gravel, or hard clean sand, with interstices or irregular surfaces to “collect” eggs and provide protection from predators, and free of excessive silt and debris that could smother eggs during incubation), larval development (*e.g.*, substrates with interstices or voids providing refuge from predators and from high flow conditions), and subadults and adults (*e.g.*, substrates for holding and spawning). Green sturgeon eggs are found in pockets of sand and gravel (2.0 to 64.0 mm in size) and in the interstitial spaces of larger substrate, such as cobble and boulders (Poytress *et al.* 2011a). Eggs are likely to adhere to sand and gravel after settling into spaces between larger substrates (Van Eenennaam *et al.* 2001b, Deng *et al.* 2002a). Larvae utilize benthic structure (Van Eenennaam *et al.* 2001b, Deng *et al.* 2002a, Kynard *et al.* 2005) and seek refuge within crevices, but will forage over hard surfaces (Nguyen and Crocker 2006). The creation of upstream dams and impoundments can reduce sediment delivery to rivers, bays and estuaries and impact the quality and quantity of spawning substrates (Draft GSRP 2016). The degree to which green sturgeon spawning habitats have been impacted in California Central Valley is not well understood but we would expect an impact commiserate with the demonstrated impacts to listed salmonid spawning habitats.

1.4.2.3 Water Flow

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include water flow - a flow regime (*i.e.*, the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages. Sufficient flow is necessary to reduce the incidence of fungal infestations of eggs, to flush fine material from feeding and rearing substrates, and to facilitate access to spawning grounds for spawning adults. On the Sacramento River, flow regimes are largely dependent on releases from Shasta Dam, thus the operation of this dam could have profound effects upon sDPS green sturgeon habitat. The majority of adult outmigration is thought to occur in the fall months when flows increase. Heublein *et al.* (2008) found that some tagged individuals out-migrated in the fall, and timing was correlated with the first winter pulse flow. However, others out-migrated in the late summer in which no known flow or temperature-related cues could be correlated. nDPS green sturgeon have exhibited similar behavior. In the Rogue River, adult green sturgeon have been shown to emigrate to the ocean during the autumn and winter when water temperatures dropped below 10°C and flows increased (Erickson *et al.* 2002). On the Klamath River, the fall outmigration of green sturgeon has been shown to coincide with a significant increase in discharge resulting from the onset of the rainy season (Benson *et al.* 2007b).

Within bays and estuaries adjacent to the Sacramento River (*i.e.*, the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary allowa adults to successfully orient to the incoming flow and migrate upstream to spawning grounds. Nakamoto *et al.* (1995a) found that juvenile growth in green sturgeon is associated with downstream migration. Adequate flows are also likely required to facilitate downstream migratory behavior in juveniles. Water flows in the estuary has been altered by channel control structures, impoundments, and upstream diversions, which have changed flow patterns, channel morphology and water depth/presence and salinity in certain areas (Draft GSRP 2016). These changes have likely impacted habitat quality, migration and movement of juvenile, subadult, and adult green sturgeon, although the extent and magnitude of impact is uncertain (Draft GSRP 2016).

In the Columbia River basin, impoundments holding water back in the summer months significantly alter water flows throughout the estuary, especially at low tide when sDPS is known to congregate there (Lindley et al. 2008, 2011). Seasonally reduced flows can alter saltwater intrusion and create salinity levels unsuitable to green sturgeon; the Columbia River estuary is impacted by saltwater intrusion more than other bays and estuaries within the range of sDPS (Draft GSRP 2016).

1.4.2.4 Water Quality

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages. Suitable water temperatures, salinities, and dissolved oxygen levels are discussed in detail in the life history section.

Summer water temperatures in the upper Sacramento River have typically ranged between or below 15 to 19^{o}C, which is within the lab-based optima for green sturgeon egg development and below lab-based optima for larval and juvenile growth (Van Eenennaam et al. 2005; Mayfield and Cech 2004; Allen et al. 2006). Notably, the water temperatures in the Sacramento River were substantially higher than these “optima” during the drought of 2014 and 2015; the impacts to green sturgeon from these higher temperatures are not well understood.

Salinity in the Sacramento River is projected to increase by 33% on average in the 21st century and water temperatures could also increase (CH2MHill 2014). These changes will result in declining habitat quality and food web productivity for green sturgeon. Laboratory experiments confirm the potential negative impacts to green sturgeon from salinity and prey base changes predicted for the San Francisco Bay Delta (Sardella and Kulz 2014; Haller et al. 2015; Vaz et al. 2015).

Green sturgeon are exposed to non-point and point source contaminants in the Sacramento River from agriculture runoff, urban development, discharge from industry and legacy contaminants from mining activities. In addition, land use practices continue to deposit mercury, heavy metals, polychlorinated biphenyls and organochlorine pesticides throughout Central Valley watersheds (Draft GSRP 2016). Contaminants currently found in the Sacramento River pose a threat to several life stages of green sturgeon: 1) eggs, larvae and juveniles resulting in reduced growth, injury or mortality, and 2) female adults during spawning resulting in negative reproductive capacity (Draft GSRP 2016).

The PBFs of sDPS green sturgeon critical habitat in estuarine habitats include water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages. Altered water temperatures are primarily a concern for the Columbia River Estuary as the other coastal bays and estuaries are not as influenced by input from large rivers with impoundments (Draft GSRP 2016). The Columbia River estuary is impacted by saltwater intrusion more than other bays and estuaries within the range of sDPS (Draft GSRP 2016). Non-point source contaminants enter the San Francisco Bay Estuary as runoff from urban sites, forests, agricultural lands, landfills, pastures, mines, nurseries, wastewater treatment, etc. and have the potential to impact juvenile growth and reproductive capacity of females (Draft GSRP 2016).

The PBFs of sDPS green sturgeon critical habitat in nearshore coastal marine areas include water quality - nearshore marine waters with adequate dissolved oxygen levels and acceptably low levels of contaminants (*e.g.,* pesticides, organochlorines, elevated levels of heavy metals) that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon. Not a lot is known about the marine habitat usage of green sturgeon or the water quality conditions in those areas.

1.4.2.5 Migratory Corridor

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include migratory corridor - a migratory pathway necessary for the safe and timely passage of Southern DPS fish within riverine habitats and between riverine and estuarine habitats (*e.g.,* an unobstructed river or dammed river that still allows for safe and timely passage). Safe and unobstructed migratory pathways are necessary for adult green sturgeon to access spawning habitats and for larval and juvenile green sturgeon to migrate downstream from spawning/rearing habitats in freshwater rivers to estuarine rearing habitats. This PBF is highly degraded compared to its historical condition because of man-made barriers and alteration of habitat. The Anderson-Cottonwood Irrigation District (ACID) Dam, at river mile (RM) 297, forms a barrier to any potential sturgeon migration. Downstream of this point, good spawning and rearing habitat exists, primarily in the river reach between Keswick Dam and Red Bluff Diversion Dam (RBDD) (RM 242). The Feather River and Yuba River also offer potential green sturgeon spawning habitat, but those rivers contain their own man-made barriers to migration and are highly altered environments.

The PBFs of sDPS green sturgeon critical habitat in estuarine habitats include migratory corridor - a migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats. sDPS green sturgeon are known to use the Sacramento River and the Sacramento-San Joaquin Delta as a migratory corridor. Additionally, certain bays and estuaries throughout Oregon and Washington and into Canada are utilized for rearing and holding, and these areas must also offer safe and unobstructed migratory corridors (Lindley *et al.* 2011).

Two key areas of concern are the Yolo and Sutter bypasses. These leveed floodplains are engineered to convey floodwaters of the greater Sacramento Valley and they include concrete weir structures (Fremont and Tisdale Weirs) that allow flood flows to escape into the bypass channels. Adult sturgeon are attracted to the bypasses by these high flows. The weirs can act as barriers, however, impeding fish passage. Fish can also be trapped in the bypasses as floodwaters recede (USFWS 1995, DWR 2005). Some of the weir structures include fish ladders intended to provide upstream passage for adult salmon, but have shown to be ineffective for providing upstream passage for adult sturgeon (Department of Water Resources and Bureau of Reclamation 2012). Also, there are irregularities in the splash basins at the foot of these weirs and multiple road crossings and agricultural impoundments in the bypasses that block hydraulic connectivity, further impeding fish passage. As a result, sturgeon may become stranded in the bypasses, delaying migration. They also may face lethal and sub-lethal effects from poaching, high water temperatures, low dissolved oxygen, and desiccation.

The PBFs of sDPS green sturgeon critical habitat in nearshore coastal marine areas include migratory corridor - a migratory pathway necessary for the safe and timely passage of Southern DPS fish within marine and between estuarine and marine habitats. There are no physical marine barriers or barriers between marine and estuarine habitats that prevent green sturgeon from migrating. Poor water quality conditions, such as anoxic conditions or acidified pulp mill effluent in the Columbia River estuary, may prevent or delay green sturgeon migration into and out of estuarine habitats but the extent of this impact is unknown (Draft GSRP 2016).

1.4.2.6 Water Depth

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include water depth - deep (≥5 m) holding pools for both upstream and downstream holding of adult or subadult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or subadult fish. Deep pools (greater than 5m depth) are critical for adult green sturgeon spawning and for summer holding within the Sacramento River. Summer aggregations of green sturgeon have been observed in deep pools above the Glen Colusa Irrigation District (GCID) diversion in the Sacramento River. The significance and purpose of these aggregations are unknown, but may be a behavioral characteristic of green sturgeon occurring elsewhere in the Delta and Sacramento River. Approximately 54 pools with adequate depth have been identified in the Sacramento River above the GCID location (Thomas *et al.* 2013). Adult green sturgeon in the Klamath and Rogue rivers also occupy deep holding pools for extended periods of time, presumably for feeding, energy conservation, and/or refuge from high water temperatures (Erickson *et al.* 2002, Benson *et al.* 2007b).

The PBFs of sDPS green sturgeon critical habitat in estuarine habitats include depth - a diversity of depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages. Habitat complexity is necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages. Subadult and adult green sturgeon occupy deep (more than 5 m) holding pools within bays, estuaries, and freshwater rivers. These deep holding pools may be important for feeding and energy conservation, or may serve as thermal refugia (Benson *et al.* 2007a). Tagged adults and subadults within the San Francisco Bay estuary primarily occupied waters with depths of less than 10 meters, either swimming near the surface or foraging along the bottom (Kelly *et al.* 2007). In a study of juvenile green sturgeon in the Delta, relatively large numbers of juveniles were captured primarily in shallow waters from 3-8 feet deep, indicating juveniles may require shallower depths for rearing and foraging (Radtke 1966b).

1.4.2.7 Sediment Quality

The PBFs of sDPS green sturgeon critical habitat in freshwater riverine systems include sediment quality (*i.e.*, chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of contaminants [*e.g.*, elevated levels of heavy metals (*e.g.*, mercury, copper, zinc, cadmium, and chromium); selenium; polycyclic aromatic hydrocarbons (PAHs); and organochlorine pesticides] that can result in negative effects on any life stage of green sturgeon and/or their prey. Metals have been shown to bio-accumulate in *Acipenserids* (taxonomic family containing green sturgeon), although less is known about its effects on their behavior at any given life stage (Kruse and Scarnecchia 2002). PAHs found in oil-based products are known to bio-accumulate in fish and have carcinogenic, mutagenic and cytotoxic effects (Johnson *et al.* 2002).

The PBFs of sDPS green sturgeon critical habitat in estuarine habitats include sediment quality (*i.e.,* chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of contaminants (*e.g.,* elevated levels of selenium, heavy metals, PAHs, and organochlorine pesticides) that can cause negative effects on all life stages of green sturgeon. Poor agricultural practices in and around the estuary result in a lowered ability for the soil to hold water which causes high run-off rates of pesticides, petroleum hydrocarbons, and other contaminants during rains events. Because these contaminants have increased permanence in the estuarine environment holding within the sediment, they likely impact green sturgeon through uptake of these contaminants when feeding. Bioaccumulation of contaminants in white sturgeon is well documented (Feist et al. 2005) and because green sturgeon occupy the same habitats and share the same prey, contaminant bioaccumulation is also likely occurring in green sturgeon (Draft GSRP 2016).

1.4.3 Green Sturgeon Life History

1.4.3.1 General Information

Green sturgeon belong to the family *Acipenseridae,* an ancient lineage of fish with a fossil record dating back approximately 200 million years. They are known to be long lived; green sturgeon captured in Oregon have been aged up to 52 years old, using a fin-spine analysis (Farr and Kern 2005). Green sturgeon are highly adapted to benthic environments, spending the majority of their lifespan residing in bays, estuaries, and near coastal marine environments. They are anadromous, migrating into freshwater riverine habitats to spawn; and iteroparous as individuals are able to spawn multiple times throughout their lifespan. Further details of their life history can be found in various literature sources such as Moyle (2002a), Adams *et al.* (2007), Beamesderfer *et al.* (2007), and Israel and Klimley (2008a).

A general timeline of green sturgeon development is given in Table 1-5. There is considerable variability across categories, such as size or age at maturity.

1.4.3.2 Adult Migration and Spawning

Green sturgeon reach sexual maturity between 15–17 years old (Beamesderfer *et al.* 2007).

Based on data from acoustic tags (Heublein *et al.* 2008), adult sDPS green sturgeon leave the ocean and enter San Francisco Bay between January and early May. Migration through the bay/Delta takes about one week and progress upstream is fairly rapid to their spawning sites (Heublein *et al.* 2008). The majority of adult green sturgeon abundance occurs in the Sacramento River, suggesting that the majority of spawning activity occurs there as well. In a recent survey, three observed sites on the Sacramento River accounted for over 50 percent of observed green sturgeon spawning (Mora, ongoing research). However, in 2011, spawning was confirmed in the Feather River by the California Department of Water Resources (Seesholtz *et al.* 2014) and suggested in the Yuba River (Bergman *et al.* 2011). Spawning activity is concentrated in the mid-April to mid-June time period (Poytress *et al.* 2013). Figure 1-15 indicates known spawning locations on the Sacramento River.

Various studies of spawning site characteristics (Poytress *et al.* 2011a) agree that spawning sDPS green sturgeon typically favor deep, turbulent holes over 5 meters deep, featuring sandy, gravel, and cobble type substrates. Spawning depth may be variable, however, for spawning has been documented in depths as shallow as 2 meters (Poytress *et al.* 2011a). Substrate type is likely constrained as the interstices of the cobble and gravel catch and hold eggs, allowing them to incubate without being washed downstream. Under laboratory conditions, green sturgeon larvae (0-15 days post hatch [DPH]) have been shown to utilize cobble and gravel for shelter, even after commencing exogenous feeding (Kynard *et al.* 2005). Adequate flows are required to create the deep, turbulent habitat that green sturgeon favor for spawning. Successful egg development requires a water temperature range between 11° and 19°C. As larvae and juveniles mature, their range of temperature tolerance increases (Table 1-6).

Table 1-5. General green sturgeon life history from egg to adult including length-life stage information.

Timeline	Life stage, Length-age relationship
Fertilization of eggs (spawning)	Spawning occurs primarily in deep water (> 5m) pools[1] at very few select sites[2], predominantly in the Sacramento River, predominantly in time period mid-April to mid-June[3]
144-192 hours (6-8 days) after fertilization of eggs	Newly hatched larvae emerge. Larvae are 12.6-14.5 mm long[4]
6 days post hatch (dph)	Nocturnal swim up, hide by day behavior observed[4]
10 dph	Exogenous feeding begins between 10-15 dph[4]. Larvae begin to disperse downstream
2 weeks old	Larvae appear in rotary screw traps at the RBDD at lengths of 24 to 31 mm.
45 dph	Larval to juvenile metamorphosis complete. Begin juvenile life stage. Juveniles are 63-94 mm in length.
45 days to 1.5 years	Juveniles migrate downstream and into the Delta or the estuary and rear to the sub-adult phase. Juveniles range in size from around 70 mm to 90 cm. Little information available about this life stage.
1.5 – 4 years	Juveniles migrate to sea for the first time, thereby entering the sub-adult phase. Subadults are 91 to 149 cm.
1.5 years to 15-17 years	Subadults enter the ocean where they grow and develop, reaching maturity between 15-17 years old*
15-17 years*	Green sturgeon reach sexual maturity and become adults, with males maturing around 120 cm and females maturing around 145 cm[5]
15 years to 50+ years	Green sturgeon have a lifespan that can reach 50 or more years and can grow to a total length of over 2 meters

Sources:

1. Thomas *et al.* (2013) 2. Mora unpublished data. 3. Poytress *et al.* (2013) 4. Deng *et al.* (2002) 5. Nakamoto *et al.* 1995 *Green sturgeon in the Klamath River might reach sexual maturity as early as 13 years for females and 9 years for males. More research is needed to determine the typical age and size of sDPS green sturgeon at maturity.

Green sturgeon fecundity is approximately 50,000–80,000 eggs per adult female (Van Eenennaam *et al.* 2001a), and they have the largest egg size of any sturgeon. The outside of the eggs are mildly adhesive and are denser than those of white sturgeon (Kynard *et al.* 2005, Van Eenennaam *et al.* 2008).

Poytress *et al.* (2012a) conducted spawning site and larval sampling in the upper Sacramento River from 2008–2012 that identified a number of spawning locations (Figure 1-15). After spawning, adults have been observed to leave the system rapidly or to hold in deep pools and migrate downriver in winter after the first storms. From 2002 to 2004 Benson *et al.* (2007a) conducted a study in which 49 adult green sturgeon were tagged with radio and/or sonic telemetry tags and tracked manually or with receiver arrays. Tagged individuals exhibited four movement patterns: upstream spawning migration, spring outmigration to the ocean, or summer holding, and outmigration after summer holding. sDPS green sturgeon that hold over the summer typically re-enter the ocean from November through January (Lindley *et al.* 2008). Benson *et al.* (2007b) also observed outmigration to the ocean in the spring.

1.4.3.3 Juvenile Migration

Larval green sturgeon hatch in the late spring or summer (peak in July) (Adams 2002) and presumably progress downstream towards the Delta as they develop into juveniles. It is uncertain when juvenile green sturgeon enter the Delta or how long they rear before entering the ocean. Ocean entry marks the transition from juvenile to sub-adults.

1.4.3.4 Egg and Larval Stages

Green sturgeon larvae have been observed hatching from fertilized eggs after approximately 169 hours at a water temperature of 15°C (59°F) (Van Eenennaam *et al.* 2001b, Deng *et al.* 2002a). Studies conducted at the University of California, Davis (UC Davis) by Van Eenennaam *et al.* (2005) indicated that an optimum range of water temperature for egg development ranged between 14°C (57.2°F) and 17.5°C (62.6°F). Eggs incubated at water temperatures between 17.5°C (63.5°F) and 22°C (71.6°F) resulted in elevated mortalities and an increased occurrence of morphological abnormalities in those eggs that did hatch (Van Eenennaam *et al.* (2005). Temperatures over 23°C (73.4°F) resulted in 100 percent mortality of fertilized eggs before hatching (Van Eenennaam (2005). Further research is needed to identify the lower temperature limits for eggs and larvae. Table 1-6 shows temperature tolerance by life stage for all stages of green sturgeon development.

Information about the life history and behavior of larval sDPS green sturgeon in the wild is very limited. The U.S. Fish and Wildlife Service (USFWS) conducts annual sampling for eggs and larvae in the mainstem Sacramento River. Larval green sturgeon appear in USFWS rotary screw traps at the RBDD from May through August (Poytress *et al.* 2010) at lengths ranging from 24 to 31 mm fork length, indicating they are approximately two weeks old (California Department of Fish and Game 2002b, U.S. Fish and Wildlife Service 2002).

This data provides limited information about green sturgeon larvae including time and date of capture and corresponding river conditions such as temperature and flow parameters.

Little is known about diet, distribution, and outmigration timing of larvae. Laboratory studies have provided some information about larval behavior, but the relevance to in-situ behavior is unknown.

Table 1-6. Green sturgeon temperature tolerance range by life stage.

temperature °C	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28
temperature °F	46.4	48.2	50.0	51.8	53.6	55.4	57.2	59.0	60.8	62.6	64.4	66.2	68.0	69.8	71.6	73.4	75.2	77.0	78.8	80.6	82.4
egg				b	b	b	b	b	b	b	b	b	b	b	b,f	b,f	b,f	b,f	b,f	b	b
larvae							e	e	e	c	f	dd,f	dd,f	dd,f	dd,f	dd,f	dd,f	dd,c,f	f	f	f
juvenile				a	a	a	a	a	a	a	a	a	a	a	a	a	a,d	a	a	a	a
spawning adult			g	g	g	g	g	g	g,h	g,h											

- optimal temperature
- acceptable temperature
- impaired fitness; avoid prolonged exposure; increasing chance of lethal effects
- likely lethal
- lethal
- unknown effect upon survival and fitness

NOTES: Life stage definitions can be found within the life stage sections of this report. Lab studies involving nDPS green sturgeon from Klamath River broodstock (a, b, c, d, dd, f) were used to rate water temperatures for the eggs, larvae, and juveniles. Water temperatures recorded during sDPS green sturgeon egg and larvae collection on the upper Sacramento and Feather rivers (e,g, and h) were used to establish 'acceptable temperature' for spawning adults and larvae.

a = Mayfield and Cech 2004
b = Van Eenennaam *et al.* 2005
c = Werner *et al.* 2006
d = Allen *et al.* 2006a
e = Poytress *et al.* 2012
f = Linares-Casenave *et al.* 2013
g = Poytress *et al.* 2015
h = Seesholtz *et al.* 2014
dd = Allen *et al.* 2006b

The figure below shows green sturgeon spawning locations in the Sacramento River from 2008-2012. [Source: Poytress *et al.* (2012b)]. Unconfirmed sites indicate an area where sturgeon have been known to congregate, but where evidence of spawning was not obtained in the study.

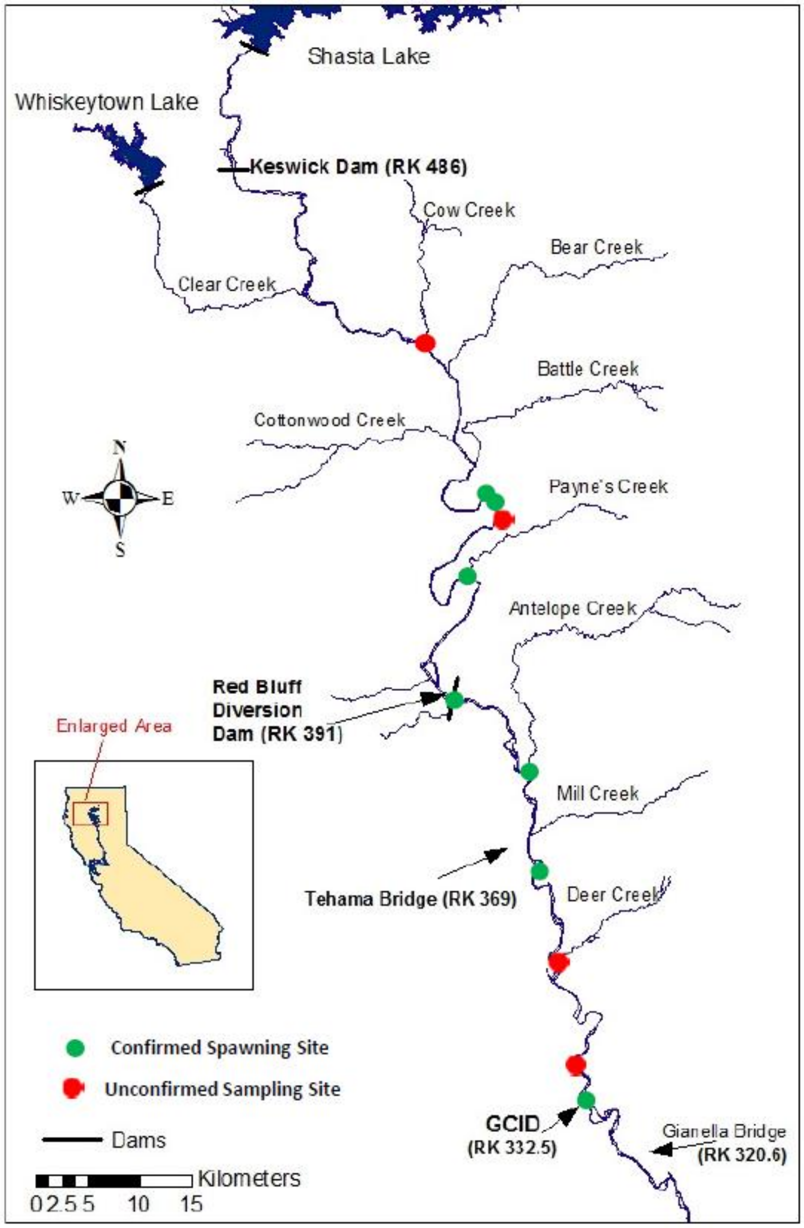


Figure 1-15. Green sturgeon spawning locations in the Sacramento River from 2008–2012

1.4.3.5 Juvenile Development and Outmigration

Juvenile green sturgeon are defined as individuals that have completed metamorphosis or are greater than 45 DPH according to Deng *et al.* (2002b). They appear to spend their first one to two months rearing in the Sacramento River (California Department of Fish and Game 2002a). Little is known about juvenile growth rates in the sDPS. Juvenile sDPS green sturgeon have been salvaged at the Federal and State pumping facilities in the southern region of the Delta and collected in sampling studies by California Department of Fish and Wildlife (CDFW) during all months of the year (California Department of Fish and Game 2002a). Salvage data have been updated through 2015, and the majority of juveniles were between 200 and 500 mm (Figure 1-16). It is important to note that few have been sampled there since 2001, and that sampling has only occurred during high water years. USWFS has sampled juvenile green sturgeon in the mainstem Sacramento River and found that some individuals reach approximately 300 mm total length (TL) in 6 months (W. Poytress, USFWS, unpublished data). The lack of any records of juveniles smaller than approximately 200 mm in the Delta may suggest smaller individuals rear in the Sacramento River or its tributaries. Juvenile green sturgeon captured in the Delta by Radtke (1966b) ranged in size from 200-580 mm, supporting the hypothesis that juvenile green sturgeon enter the Delta after 10 months or when they are greater than 200 mm in size.

Radtke (1966b) inspected the stomach contents of juvenile green sturgeon (range: 200-580 mm) in the Delta and found food items to include mysid shrimp (*Neomysis awatschensis*), amphipods (*Corophium sp.*), and other unidentified shrimp. In the northern estuaries of Willapa Bay, Grays Harbor, and the Columbia River, green sturgeon have been found to feed on a diet consisting primarily of benthic prey and fish common to the estuary. For example, burrowing thalassinid shrimp (mostly *Neotrypaea californiensis*) were important food items for green sturgeon taken in Willapa Bay, Washington (Dumbauld *et al.* 2008).

1.4.3.6 Estuarine Rearing

The age of first ocean entry in sDPS green sturgeon is poorly understood. Juvenile green sturgeon in the nDPS may spend 2 to 3 years in fresh or brackish water before making their first migration to sea. Nakamoto *et al.* (1995b) found that, on average, green sturgeon on the Klamath River migrated to sea by age three and no later than age four. On the Klamath River (nDPS), Allen *et al.* (2009) devised a technique to estimate the timing of transition from fresh water to seawater by taking a bone sample from the leading edge of the pectoral fin and analyzing the strontium/calcium ratios. The results of this study indicate that nDPS green sturgeon move from freshwater to brackish water at 0.5–1.5 years old and then move into seawater at 2.5-3.5 years old. Moyle (2002a) suggests that sDPS green sturgeon migrate out to sea before the end of their second year and perhaps as young of the year (YOY). Laboratory experiments indicate that green sturgeon juveniles may occupy fresh to brackish water at any age, but they gain the physiological ability to transition to saltwater at approximately 1.5 years old (Allen and Cech 2007).

1.4.3.7 Ocean Rearing

Once green sturgeon juveniles make their first entry into sea, they enter the sub-adult phase and spend multiple years migrating along the coastal zones, bays, and estuaries (Lindley *et al.* 2008). Sub-adult green sturgeon have not been observed in freshwater spawning areas. Green sturgeon mature at approximately 15 to 20 years old, and an individual may spawn once every 2-4 years and live for 50 years or more (Moyle 2002a, Israel and Klimley 2008b).

In the summer months, multiple rivers and estuaries throughout the sDPS range are visited by dense aggregations of adult green sturgeon (Moser and Lindley 2006, Lindley *et al.* 2011). Genetic studies on green sturgeon stocks indicate that the green sturgeon in the San Francisco Bay ecosystem belong exclusively to the sDPS (Israel *et al.* 2009). Capture of green sturgeon as well as tag detections in tagging studies have shown that green sturgeon are present in San Pablo Bay and San Francisco Bay at all months of the year (Kelly 2007, Heublein *et al.* 2008, Lindley *et al.* 2011). An increasing amount of information is becoming available regarding green sturgeon habitat use in estuaries and coastal ocean and why they aggregate episodically (Lindley *et al.* 2008, Lindley *et al.* 2011).

1.4.4 Green Sturgeon Viable Salmonid Population Parameters

As an approach to determining the conservation status of salmonids, NMFS has developed a framework for identifying attributes of a viable salmonid population (VSP). The intent of this framework is to provide parties with the ability to assess the effects of management and conservation actions and to ensure their actions promote the listed species' survival and recovery. This framework is known as the VSP concept (McElhany *et al.* 2000a). The VSP concept measures population performance in terms of four key parameters: abundance, population growth rate, spatial structure, and diversity. Although the VSP concept was developed for Pacific salmonids, the underlying parameters are general principles of conservation biology and can therefore be applied more broadly. Here, we adopt the VSP parameters for analyzing sDPS green sturgeon viability.

1.4.4.1 Abundance

Trends in abundance of sDPS green sturgeon have been estimated from two long-term data sources:

(1) salvage numbers at the State and Federal pumping facilities (see below),

(2) by incidental catch of green sturgeon by the CDFW's white sturgeon sampling/tagging program.

Historical estimates from these sources are likely unreliable as sDPS green sturgeon were likely not taken into account in incidental catch data, and salvage does not capture range-wide abundance in all water year types. Recently, more rigorous scientific inquiry has been undertaken to generate abundance estimates (Israel and May 2010, Mora *et al.* 2015).

A decrease in sDPS green sturgeon abundance has been inferred from the amount of take observed at the south Delta pumping facilities: the Skinner Delta Fish Protection Facility (SDFPF) and the Tracy Fish Collection Facility (TFCF). This data should be interpreted with some caution; operations and practices at the facilities have changed over the decades, which may affect the salvage data shown below (Figure 1-16). The salvage data likely indicate a high production year versus a low production year qualitatively, but cannot be used to rigorously quantify abundance. Despite the potential pitfalls of using salvage data to estimate trends in abundance for sDPS green sturgeon, Figure 1-16 indicates a steep decline in abundance.

Since 2010, more robust estimates of sDPS green sturgeon have been generated. As part of a doctoral thesis at UC Davis, Ethan Mora has been using acoustic telemetry as well as DIDSON (dual-frequency identification sonar) to locate green sturgeon in the Sacramento River and to derive an adult spawner abundance estimate (Mora *et al.* 2015). Results of these surveys estimate an average annual spawning run of 223 (DIDSON) and 236 (telemetry) fish. This estimate does not include the number of spawning adults in the lower Feather River, where green sturgeon spawning was recently confirmed (Seesholtz *et al.* 2014).

The image below shows annual salvage of green sturgeon for the SDFPF and the TFCF 1981–2015. Data source: http://www.dfg.ca.gov/delta/apps/salvage/Default.aspx

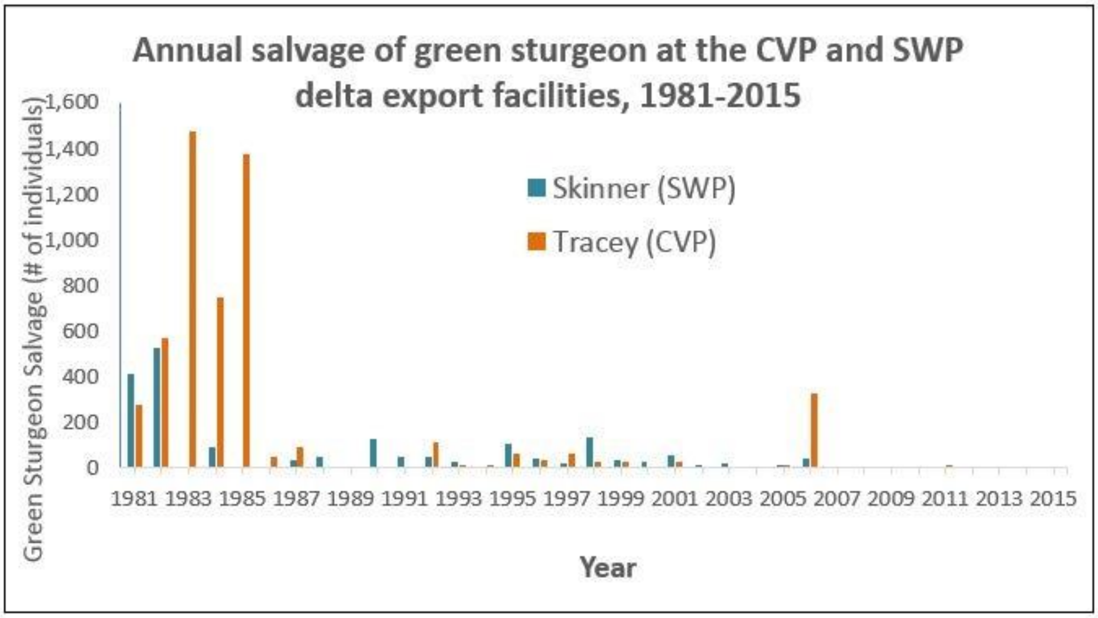


Figure 1-16. Annual salvage of green sturgeon for the SDFPF and the TFCF 1981–2015.

1.4.4.2 Productivity

The parameters of green sturgeon population growth rate and carrying capacity in the Sacramento Basin are poorly understood. Larval count data are available from rotary screw traps set seasonally near Red Bluff and Glen Colusa irrigation diversions. This data shows enormous variance among years with the greatest number of larval green sturgeon occurring in 2011 when 3,700 larvae were captured (Poytress *et al.* 2012b). In other years, larval counts were an order of magnitude lower. In general, sDPS green sturgeon year class strength appears to be highly variable with overall abundance dependent upon a few successful spawning events (National Marine Fisheries Service 2010). Other indicators of productivity such as data for cohort replacement ratios and spawner abundance trends are not currently available for sDPS green sturgeon. The long lifespan of the species and long age to maturity makes trend detection dependent upon data sets spanning decades. The acoustic telemetry work begun by Mora (UC Davis) on the Sacramento River and by Seesholtz *et al.* (2014) (CDWR) on the Feather River, as well as larval and juvenile studies by Poytress *et al.* (2011b) (USFWS), may eventually produce a more statistically robust analysis of productivity.

1.4.4.3 Spatial Structure

Green sturgeon are known to range from Baja California to the Bering Sea along the North American continental shelf. During late summer and early fall, subadults and non-spawning adult green sturgeon can frequently be found aggregating in estuaries along the Pacific coast (Emmett *et al.* 1991, Moser and Lindley 2006). Using polyploid microsatellite data, Israel *et al.* (2009) found that green sturgeon within the Central Valley of California belong to the sDPS. Additionally, acoustic tagging studies have found that green sturgeon found spawning within the Sacramento River are exclusively sDPS green sturgeon (Lindley *et al.* 2011).

In waters inland from the Golden Gate Bridge in California, sDPS green sturgeon are known to range through the estuary and the Delta and up the Sacramento, Feather, and Yuba rivers (Isreal et al. 2009, Cramer Fish Sciences 2011, Seesholtz et al. 2014). The minimum northern-most extent of this range is thought to be Cow Creek (Mora, unpublished data). In the Yuba River, green sturgeon have been documented up to Daguerre Point Dam (Bergman *et al.* 2011), which currently impedes access to areas upriver. Similarly, in the Feather River, green sturgeon have been observed by CDWR staff up to the Fish Barrier Dam. On the Sacramento River, the ACID dam at RM 297 is thought to be the highest point on the river accessible to green sturgeon. Viable spawning habitat may exist up to this point (Seesholtz 2015). Adult green sturgeon were detected up the confluence with Cow Creek (River Kilometer [RK]) 450) in 2005, and spawning was confirmed at the confluence with Ink's Creek (RK 426) in 2011 (Poytress *et al.* 2012a). Adams *et al.* (2007) summarizes information that suggests green sturgeon may have been distributed above the locations of present-day dams on the Sacramento and Feather rivers. Mora *et al.* (2009) analyzed and characterized known green sturgeon habitat and used that characterization to identify potential green sturgeon habitat within the Sacramento and San Joaquin River basins, which now lies behind impassable dams. This study concludes that approximately 9 percent of historically available habitat is now blocked by impassible dams. It is likely that this blocked habitat was of high quality for spawning.

Studies conducted at UC Davis (Mora, unpublished data) have shown that green sturgeon spawning sites are concentrated in just a handful of locations. Mora found that in the Sacramento River, just three sites accounted for over 50 percent of the green sturgeon documented in June of 2010, 2011, and 2012. This finding has important implications for the application of the spatial structure VSP parameter, which is largely concerned with spatial structuring of spawning habitat. Given the high density of individuals within a few spawning sites, extinction risk due to stochastic events is expected to have increased since the onset of dam construction and habitat loss in Central and Northern California.

Green sturgeon have been historically captured and are regularly detected within the Delta area of the lower San Joaquin River. Anglers have reported catching a small number of green sturgeon at various locations in the San Joaquin River upriver of the Delta (Gleason et al. 2008; DuBois et al. 2009, 2010, 2011, 2012). However, there is no known modern usage of the upper San Joaquin River, and adult green sturgeon spawning has not been documented (Jackson and Eenennaam 2013). Based on this information, it is unlikely that green sturgeon utilize areas of the San Joaquin River upriver of the Delta with regularity, and spawning events are thought to be limited to the upper Sacramento River and its tributaries.

Recent research indicates that the sDPS is composed of a single, independent population, which principally spawns in the mainstem Sacramento River (Isreal et al. 2009), and also breeds opportunistically in the Feather River and possibly even the Yuba River (Cramer Fish Sciences 2011; Seesholtz et al. 2014). Concentration of adults into a very few select spawning locations makes the species highly vulnerable to poaching and catastrophic events. The apparent, but unconfirmed, extirpation of spawning populations from the San Joaquin River narrows the available habitat within their range, offering fewer habitat alternatives.

1.4.4.4 Diversity

Diversity, as defined in the VSP concept in (McElhany *et al.* 2000a), includes purely genetically driven traits such as DNA sequence variation, as well as traits that are driven by a combination of genetics and the environment such as ocean behavior, age at maturity, and fecundity.

Variation is important to the viability of a species for several reasons. First, it allows a species to utilize a wide array of environments. Second, diversity protects a species from short-term spatial and temporal changes in the environment by increasing the likelihood that at least some individuals will persist in spite of changing environmental conditions. Third, genetic diversity facilitates adaptation to changing environmental conditions over the long term.

Whether sDPS green sturgeon display these diversity traits and if there is sufficient diversity to buffer against long term extinction risk is not well understood. It is likely that the diversity of sDPS green sturgeon is low, given recent abundance estimates. Human alteration of the environment is pervasive in the California Central Valley. As a result, many aspects of sDPS green sturgeon diversity such as run timing and behavior have likely been adversely influenced through mechanisms such as altered flow and temperature regimes.

1.4.4.5 Summary of DPS viability

The viability of sDPS green sturgeon is constrained by factors such as a small population size, lack of multiple populations, and concentration of spawning sites into just a few locations. The risk of extinction is believed to be moderate (National Marine Fisheries Service 2010). Although threats due to habitat alteration are thought to be high and indirect evidence suggests a decline in abundance, there is much uncertainty regarding the scope of threats and the viability of population abundance indices (National Marine Fisheries Service 2010). Viability is defined as an independent population having a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year timeframe (McElhany *et al.* 2000a).

Although the population structure of sDPS green sturgeon is still being refined, it is currently believed that only one population of sDPS green sturgeon exists. Lindley *et al.* (2008), in discussing winter-run Chinook salmon, states that an Evolutionarily Significant Unit (ESU) represented by a single population at moderate risk of extinction is at high risk of extinction over a large timescale. This concern applies to any DPS or ESU represented by a single population, suggesting that sDPS green sturgeon face a high extinction risk in the future. NMFS determined, upon weighing all available information (and lack of information) that the extinction risk to sDPS green sturgeon is moderate (National Marine Fisheries Service 2010).

There is a strong need for additional information about sDPS green sturgeon, especially with regards to a more robust estimate of abundance and population trends, and a greater understanding of biology and habitat needs. The most recent 5-year status review for sDPS green sturgeon found that some threats to the species have recently been eliminated, such as take from commercial fisheries and removal of some passage barriers (National Marine Fisheries Service 2015). Since many of the threats cited in the original listing still exist, the threatened status of the DPS is still applicable (National Marine Fisheries Service 2015). The 2015 5-year status review calls for the following future actions to be taken to contribute to the recovery of this species:

(1) Continue monitoring and studying key life history stages and modeling population abundance,

(2) Achieve a comprehensive understanding of annual take of Southern DPS green sturgeon, and

(3) Improve spawning habitat availability and quality (National Marine Fisheries Service 2015).

1.5 Climate Change

One major factor affecting the rangewide status of the threatened and endangered anadromous fish in the Central Valley, and aquatic habitat at large is climate change. Lindley et al. (2007) summarized several studies (Hayhoe et al. 2004, Dettinger et al. 2004, Dettinger 2005, VanRheenen et al. 2004, Knowles and Cayan 2002) on how anthropogenic climate change is expected to alter the Central Valley, and based on these studies, described the possible effects to anadromous salmonids. Climate models for the Central Valley are broadly consistent in that temperatures in the future will warm significantly, total precipitation may decline, the variation in precipitation may substantially increase (i.e., more frequent flood flows and critically dry years), and snowfall will decline significantly (Lindley et al. 2007). Climate change is having, and will continue to have, an impact on salmonids throughout the Pacific Northwest and California (Battin et al. 2007).

Warmer temperatures associated with climate change reduce snowpack and alter the seasonality and volume of seasonal hydrograph patterns (Cohen *et al*. 2000). Central California has shown trends toward warmer winters since the 1940s (Dettinger and Cayan 1995). An altered seasonality results in runoff events occurring earlier in the year due to a shift in precipitation falling as rain rather than snow (Roos 1991, Dettinger *et al*. 2004). Specifically, the Sacramento River basin annual runoff amount for April-July has been decreasing since about 1950 (Roos 1987, 1991). Increased temperatures influence the timing and magnitude patterns of the hydrograph.

The magnitude of snowpack reductions is subject to annual variability in precipitation and air temperature. The large spring snow water equivalent (SWE) percentage changes, late in the snow season, are due to a variety of factors including reduction in winter precipitation and temperature increases that rapidly melt spring snowpack (VanRheenen *et al*. 2004). Factors modeled by VanRheenen *et al*. (2004) show that the melt season shifts to earlier in the year, leading to a large percent reduction of spring SWE (up to 100% in shallow snowpack areas). Additionally, an air temperature increase of 2.1°C (3.8°F) is expected to result in a loss of about half of the average April snowpack storage (VanRheenen *et al*. 2004). The decrease in spring SWE (as a percentage) would be greatest in the region of the Sacramento River watershed, at the north end of the Central Valley, where snowpack is shallower than in the San Joaquin River watersheds to the south.

Modeling indicates that stream habitat for cold-water species declined with climate warming and remaining habitat suitable may only exist at higher elevations (Null et al 2013). Climate warming is projected to cause average annual stream temperatures to exceed 24°C slightly earlier in the spring, but notably later into August and September. The percentage of years that stream temperatures exceeded 24°C (for at least 1 week) is projected to increase, so that if air temperatures rise by 6°C, most Sierra Nevada rivers would exceed 24°C for some weeks every year.

Warming is already affecting Central Valley Chinook salmon. Because the runs are restricted to low elevations as a result of impassable rim dams, if climate warms by 5°C (9°F), it is questionable whether any Central Valley Chinook salmon populations can persist (Williams 2006). Based on an analysis of an ensemble of climate models and emission scenarios and a reference temperature from 1951- 1980, the most plausible projection for warming over

Northern California is 2.5°C (4.5°F) by 2050 and 5°C by 2100, with a modest decrease in precipitation (Dettinger 2005). Chinook salmon in the Central Valley are at the southern limit of their range, and warming will shorten the period in which the low elevation habitats used by naturally-producing Chinook salmon are thermally acceptable. This should particularly affect fish that emigrate as fingerlings, mainly in May and June, and especially those in the San Joaquin River and its tributaries.

Central Valley salmonids are highly vulnerable to drought conditions. The increased in-river water temperature resulting from drought conditions is likely to reduce the availability of suitable holding, spawning, and rearing conditions in Clear Creek, and in the Sacramento, Feather, and Yuba rivers. During dry years, the availability of thermally suitable habitats in spring-run Chinook salmon river systems without major storage reservoirs (e.g., Mill, Deer, and Butte creek) is also likely to be reduced. Multiple dry years in a row could potentially devastate Central Valley salmonids. Prolonged drought due to lower precipitation, shifts in snowmelt runoff, and greater climate extremes could easily render most existing spring-run Chinook salmon habitat unusable, either through temperature increases or lack of adequate flows. The drought that occurred from 2007-2009 was likely a factor in the recent widespread decline of all Chinook salmon runs (including spring-run Chinook) in the Central Valley (Williams et al. 2011).

The increase in the occurrence of critically dry years also would be expected to reduce abundance, as, in the Central Valley, low flows during juvenile rearing and outmigration are associated with poor survival (Kjelson and Brandes 1989, Baker and Morhardt 2001, Newman and Rice 2002). In addition to habitat effects, climate change may also impact Central Valley salmonids through ecosystem effects. For example, warmer water temperatures would likely increase the metabolism of predators, reducing the survival of juvenile salmonids (Vigg and Burley 1991). In summary, climate change is expected to exacerbate existing stressors and pose new threats to Central Valley salmonids, including the CV spring-run Chinook, by reducing the quantity and quality of inland habitat (Lindley et al. 2007).

Over the last five years, there has been a period of widespread decline in all Central Valley Chinook salmon stocks. An analysis by Lindley et al. (2009) that examined fall-run Chinook found that unusual oceanic conditions led to poor growth and survival for juvenile salmon entering the ocean from the Central Valley during the spring of 2005 and 2006 and most likely contributed to low returns in 2008 and 2009. This reduced survival was attributed to weak upwelling, warm sea surface temperatures, low prey densities, and poor feeding conditions in the ocean. When poor ocean conditions are combined with drought conditions in the freshwater environment the productivity of salmonid populations can be significantly reduced. Although it is unclear how these unusual ocean conditions affected CCV steelhead, it is highly likely they were adversely impacted by a combination of poor ocean conditions and drought over the past five years (National Marine Fisheries Service 2011).

For Sacramento River winter-run Chinook salmon, the embryonic and larval life stages that are most vulnerable to warmer water temperatures occur during the summer, so this run is particularly at risk from climate warming. The only remaining population of winter-run Chinook salmon relies on the cold water pool in Shasta Reservoir, which buffers the effects of warm temperatures in most years. The exception occurs during drought years, which are predicted to occur more often with climate change (Yates et al. 2008). The long-term projection of how the CVP/SWP will operate incorporates the effects of potential climate change in three possible forms: less total precipitation; a shift to more precipitation in the form of rain rather than snow; or, earlier spring snow melt (Reclamation 2008). Additionally, air temperature appears to be increasing at a greater rate than what was previously analyzed (Lindley 2008, Beechie *et al*. 2012, Dimacali 2013). These factors will compromise the quantity and/or quality of winter-run Chinook salmon habitat available downstream of Keswick Dam. It is imperative for additional populations of winter-run Chinook salmon to be re-established into historical habitat in Battle Creek and above Shasta Dam for long-term viability of the ESU (National Marine Fisheries Service 2014a).

Spring-run Chinook salmon adults are vulnerable to climate change because they over-summer in freshwater streams before spawning in autumn (Thompson et al. 2011). CV spring-run Chinook salmon spawn primarily in the tributaries to the Sacramento River, and those tributaries without cold water refugia (usually input from springs) will be more susceptible to impacts of climate change. Even in tributaries with cool water springs, in years of extended drought and warming water temperatures, unsuitable conditions may occur. Additionally, juveniles often rear in the natal stream for one to two summers prior to emigrating, and would be susceptible to warming water temperatures (National Marine Fisheries Service 2016b). In Butte Creek, fish are limited to low elevation habitat that is currently thermally marginal, as demonstrated by high summer mortality of adults in 2002 and 2003, and will become intolerable within decades if the climate warms as expected. Ceasing water diversion for power production from the summer holding reach in Butte Creek resulted in cooler water temperatures, more adults surviving to spawn, and extended population survival time (Mosser *et al*. 2013).

Although steelhead will experience similar effects of climate change to Chinook salmon, as they are also blocked from the vast majority of their historic spawning and rearing habitat, the effects may be even greater in some cases, as juvenile steelhead need to rear in the stream for one to two summers prior to emigrating as smolts. In the Central Valley, summer and fall temperatures below the dams in many streams already exceed the recommended temperatures for optimal growth of juvenile steelhead, which range from 14°C to 19°C (57°F to 66°F). Several studies have found that steelhead require colder water temperatures for spawning and embryo incubation than salmon (McCullough et al. 2001). In fact, McCullough et al. (2001) recommended an optimal incubation temperature at or below 11°C to 13°C (52°F to 55°F). Successful smoltification in steelhead may be impaired by temperatures above 12°C (54°F), as reported in Richter and Kolmes (2005). As stream temperatures warm due to climate change, the growth rates of juvenile steelhead could increase in some systems that are currently relatively cold, but potentially at the expense of decreased survival due to higher metabolic demands and greater presence and activity of predators. Stream temperatures that are currently marginal for spawning and rearing may become too warm to support wild steelhead populations.

Southern DPS green sturgeon spawn primarily in the Sacramento River in the spring and summer. Anderson-Cottonwood Irrigation District Diversion Dam (ACID) is considered the upriver extent of green sturgeon migration in the Sacramento River (71 FR 17757, April 7, 2006). The upriver extent of green sturgeon spawning, however, is approximately 30 kilometers downriver of ACID because water temperatures in this section of the river are too cold for spawning (Draft GSRC 2016). Thus, if water temperatures increase with climate change, temperatures adjacent to ACID may remain within tolerable levels for the embryonic and larval life stages of green sturgeon, but temperatures at spawning locations lower in the river may be more affected. It is uncertain, however, if green sturgeon spawning habitat exists closer to ACID, which could allow spawning to shift upstream in response to climate change effects. Successful spawning of green sturgeon in other accessible habitats in the Central Valley (*i.e.*, the Feather River) is limited, in part, by late spring and summer water temperatures (National Marine Fisheries Service 2015). Similar to salmonids in the Central Valley, green sturgeon spawning in tributaries to the Sacramento River is likely to be further limited if water temperatures increase and higher elevation habitats remain inaccessible.

In summary, observed and predicted climate change effects are generally detrimental to all of the species (McClure 2011, Wade et al. 2013), so unless offset by improvements in other factors, the status of the species and critical habitat is likely to decline over time. The climate change projections referenced above cover the time period between the present and approximately 2100. While there is uncertainty associated with projections, which increases over time, the direction of change is relatively certain (McClure et al. 2013).

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2015-2016 Chinook Salmon Dewatered Redd Monitoring on the Sacramento River
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